CENTRE OF EXPERTISE FOR WATERS

# Assessing the socio-economic impacts of soil degradation on Scotland's water environment

Nikki Baggaley, Fiona Fraser, Paul Hallett, Allan Lilly, Mohamed Jabloun, Kenneth Loades, Thomas Parker, Mike Rivington, Amin Sharififar, Zulin Zhang, Michaela Roberts

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## Report and Appendices

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# Acronyms

**HOST**: A hydrological classification of soils that makes use of the fact that the physical properties of soils that have a major influence on catchment hydrology (Boorman *et al.,* 1995).

**HOST SPR:** Standard Percentage runoff calculated for each soil HOST class

**NSIS1978-88:** National Soil Inventory of Scotland: Sampling was carried out on a 10 km grid between 1978 and 1988, and 721 locations were sampled. The soil and landscape at each sampling location are described and a wide range of soil properties are measured in the laboratory from soil samples taken at these locations. These include contextual information describing the surrounding landscape (such as slope and vegetation) and detailed chemical and physical analyses of each horizon (layer) within the soil profile. https://soils.environment.gov.scot/maps/point-data/national-soil-inventory-of-scotland-nsis-1978-88/

**NSIS2007-09:** National Soil Inventory of Scotland: Resampling of NSIS1978-88 on a 20 km grid between 2007 and 2009, and 183 locations were sampled to measure changes in the preceding 25 years and test new indicators of soil health including emerging contaminants https://www.hutton.ac.uk/sites/default/files/files/soils/NSIS%202007-2009%20Field%20Protocols.pdf

**Peat soil:** Is a soil type which in Scotland is defined as having a surface peat layer with more than 60% organic matter and of at least 50cm thickness.

**Peaty soils:** Have a shallower peat layer (<50cm) at the surface.

**Peatlands:** Often called bog, are wetland ecosystems with soils which are saturated and hold large deposits of dead organic material in different stage of humification (peat). In good conditions, peatland habitats are dominated by a mixture of Sphagnum mosses, cotton-grass (Eriophorum spp.), and dwarf shrubs.

**Soil compaction:** Soil compaction occurs where the soil particles and aggregates are compressed reducing pore volume and increasing soil density. It can also distort soil pores through shearing, so transmission pathways can be disrupted. Soils can be compacted in both the topsoil and subsoil.

**Soil contamination**: Soil contamination is regarded as the presence of chemicals or other substances in the soil in concentrations that may be harmful to humans, the environment or biodiversity.

**Soil sealing:** Soil sealing can be defined as the destruction or covering of the ground by an impermeable material, which is directly associated with the degree of urbanisation or transport network.

# Executive Summary

## Background

Soil degradation and soil health underpin and integrate many policy areas including parts of the Scottish National Adaptation Plan (2024–2029) and the Scottish Biodiversity Strategy. These emphasise the importance of managing soils as a vital national asset. Soils are vital in underpinning agricultural production, managing water flows and protecting water quality. Moreover, they are a major sink and source of greenhouse gases, especially by sequestering carbon from the atmosphere. The poor management of soils has the potential to lead to biodiversity decline, as well as damage to human health and built infrastructure. Understanding the value of maintaining healthy soils, especially under a changing climate, is therefore a key component in directing funding and investment into Scotland's soil assets.

## Purpose of research

This project was set out to develop a framework and collate the data available to cost the degradation of soils across Scotland. The aims of the project were to:

- Estimate the effects of soil degradation (compaction, sealing and contamination) on soil properties.
- Calculate the extent of these degradation processes.
- Assess the wider impacts on soil functions.
- Cost associated impacts of the loss of soil functioning.

## Methods

We have drawn together impact, extent, and costs to estimate the overall cost of degradation processes at the Scottish scale and identified data gaps where this could not be achieved.
This includes:

- An estimation of the extent of soil compaction in cultivated land in Scotland based on measured data.
- Estimation of changes in sealing extent using the Ordnance Survey MasterMap® data as used in the NatureScot Sealing Ecosystem Health Indicator.
- Modelling the impact of soil compaction on crop yield.
- Modelling the impact of soil compaction and soil sealing on runoff.
- A review of the data and thresholds available to assess the extent impact and costs of soil contamination.

## Key findings and recommendations

### Total costs

- Soil compaction alone already costs farmers more than £25 million per year due to yield loss and additional fuel use. The cost of worsening compaction could exceed £70 million at the farm gate, without considering more inefficient fertiliser use in compacted soils. The overall costs of soil degradation to the wider Scottish economy are far greater due to impacts from erosion, flooding, contamination, biodiversity loss and greenhouse gas production.

### Soil compaction extent

- Using available data from across Scotland we estimate that soil compaction extent on cultivated land 56% at high, 21% at moderate and 12% at low vulnerability.
    - o **Recommendation:** A national field-based assessment of the extent of both topsoil and subsoil compaction (recommended by the draft EU Soil Monitoring and Resilience Directive) to provide improved understanding of the relationships between land management intensity, erosion, runoff and compaction.

### Soil compaction: Costs of reduced crop yield

- Due to soil compaction reducing water available to plants we estimate a yield loss to spring barley of 109,000 – 324,000 tonnes depending on the proportion of land compacted. This equates to a loss of between £16 million and £49 million across Scotland per year and is likely to be exacerbated by climate change, particularly droughts in spring. There will also be associated cascading costs in terms of losses to the economy within the spring barley supply chain (e.g. livestock and distilling sectors).

- **Recommendation:** Further research to:
  - Understand soil-climate combinations and the impact of changes in nitrogen dynamics due to compaction has on yield.
  - Explore the impact of compaction on yield for other crop types.

### Soil compaction and soil sealing: Costs of runoff and flooding

- Focusing on seven catchments, we calculated catchment runoff increased by between 0.2 and 7.8 % due to compaction in cultivated land. This additional runoff is likely to increase flood risk.
- In a small sub-catchment of the River Dee with significant urban development we estimate a 1.5% increase in runoff from an additional 1% of sealed area.
- An increase in flooding by 1% could increase local authority flood damage costs by £2.6 million per year. In addition, insurance claims for flooding in Scotland are estimated at £57,000 – £76,000 per property for a single flood event.
  - **Recommendations:**
    - Align flood risk mapping with estimates of additional runoff from compaction.
    - Integrate the additional runoff estimates into SEPA flood modelling to assess increases in flood extent.
    - Estimate the number and type of properties flooded and therefore the wider costs.

### Soil compaction: Costs of decreasing water quality and nutrient replacement

- Compaction will exacerbate runoff and erosion and potentially further reduce water quality. This will increase costs of erosion above that were estimated in previous studies.
  - **Recommendation:** Develop a framework to combine impacts and costs of compaction and erosion.

### Soil compaction: Costs of additional fuel use

- Soil compaction leads to increasing fuel use, estimated to cost between £15 and £209 per hectare depending on the crop type. This translates to an additional fuel use at a national level of between £9 million and £26 million across Scotland.
  - **Recommendation:** Refine additional costs by soil type & management.

### Soil contamination: Poses risks to human and animal health.

- Contamination will impact water and food quality, as well as potentially cause damage to human and animal health and ultimately the loss of land. There are also likely to be yet unknown impacts from emerging contaminants such as microplastics and persistent organic pollutants.
  - **Recommendation:** Further research on impacts of particularly emerging contaminants on soils, their extent, and thresholds at which there is a loss of soil functions or the need for remediation.

### Soil biodiversity

- Soil Biodiversity is widely recognised as being crucial to the functioning of the soil ecosystem but also impacted by all the degradation processes reviewed. No extensive data sets on the impacts of compaction, sealing or contamination on soil biodiversity exist for Scotland.
  - **Recommendations:**
    - Establish a list of practical, useful metrics of soil biodiversity which could be monitored across Scotland.
    - Establish whether past publicly funded small-scale datasets containing these metrics can be merged to establish a baseline estimate.

# 1 Introduction

## 1.1 Background and objectives

Soil degradation and soil health are part of the Scottish National Adaptation Plan (2024–2029) which emphasises the importance of managing soils as a vital national asset. The aim of this project is to demonstrate the importance of maintaining healthy soils and estimating the costs to the nation of degrading them.

Soil degradation has impacts on both water quality and quantity as well as impacting on the soils ability to provide the ecosystem goods and services demanded by society. Options to mitigate the potential impacts of soil degradation include investment in agricultural and ecosystem services payments, flood management, peatland restoration, and sustainable soil management amongst others. Understanding the extent, spatial distribution, drivers and socio-economic costs of soil degradation and its impacts on waters, will provide a basis for prioritising and targeting investments to protect soils, mitigating detrimental impacts on soils to the wider environment, specifically water.

In discussion with the project steering group the aim of the project was to develop a robust methodological framework to assess the socio-economic and environmental impacts of three soil degradation processes on land-based businesses and the wider off-site impacts on Scotland's water environment:

- Soil compaction,
- Soil contamination and
- Soil sealing

The process of method development included appraisal of the:

- Data needs and the utility of existing data to define the EXTENT of the degradation in Scotland,
- Methods and modelling approaches available for quantifying the IMPACTS of degradation,
- Data available to COST the impacts of degradation,
- Requirements for method validation and the identification of existing knowledge, data, and evidence gaps.

This work adds to current and previous work on costings of:

- Soil erosion (Rickson *et. al.,* 2019) including the losses of sediments and nutrients to water courses as a result of soil type and land management.
- The degradation of organic soils and peatlands in the current Scottish Government Strategic Research Programme (Centre Peat: 2022–2027) which aims to define the extent, impacts and costs of the degradation of organic soils and peatlands and includes costings of dissolved organic carbon on drinking water quality.

## 1.2 Economics of soil degradation

The valuation of ecosystem services can provide easily contextualised information on the importance of preserving ecosystem function. Through measuring benefits in a single unit, in this case British Pounds (GBP), economic valuation allows for comparison of benefits between services and in a language that is readily understood by policy makers and practitioners. Although care must be taken to appreciate the holistic nature of ecosystems and the services they provide, the estimation of an economic value can prove a useful tool in decision making. In the case of ecosystem degradation, valuation of ecosystem services can make tangible what is being lost and present an argument for its conservation (Baveye *et. al.,* 2016), and promote cost-effective management (Tepes *et. al.,* 2021).

Soils have often been overlooked in valuation of ecosystem service provision (Baveye *et al.,* 2016; Jónsson and Davíðsdóttir, 2016), and are frequently bundled under "land" values (Baveye *et. al.,* 2016; Nkonya *et. al.,* 2016). Current estimates of soil value are limited in both their geography and the types of services considered (Bartkowski *et. al.,* 2020; Baveye *et al.,* 2016). Most economic research on soils tends to focus on the direct services, such as crop growth, with little consideration of aspects such as health and wellbeing (Tepes *et. al.,* 2021), or non-use values and stated preferences (Bartkowski *et al.,* 2020; Baveye *et. al.,* 2016).

Worldwide, the value of soil ecosystem services has been estimated between £2 to £23,000/ha, depending on services valued, but this has been constrained by significant gaps in data to enable full valuation (Jónsson and Davíðsdóttir, 2016). Broader degradation of land, which may extend beyond just soil impacts, has been estimated at $428bn (£336bn) (Nkonya *et. al.,* 2016).

Within the UK, soil degradation in England and Wales was estimated to cost £15.87bn per year (2023 values), based on the loss of soil organic matter (47%), erosion (12%) and compaction (39%), and with 80% of costs being experienced off-site (Graves *et al.,* 2015). In Scotland, erosion has been the most comprehensively costed service, estimated at £49.5million per year, although it is important to note that many services could not be valued due to lack of data (Rickson *et. al.,* 2019).

The value of Scotland's soil stock is realised through the services (flows) it provides. As Scotland's soils become degraded and the services that soils are able to provide diminish, soils decline in value. We can therefore estimate the cost of soil degradation as the difference in value of the services provided by soils in its pre and post degradation states (Figure 1). There is also capacity to restore or modify soils to enhance ecosystem services, leading to economic benefits, however this is outside of the scope of this research.

Costs can be categorised as on-site costs, which occur at the site of degradation, and off-site costs, which occur distant from the site of degradation (Table 1). On-site costs are typically estimated through market prices (e.g. yield, fertiliser, fuel), while off-site may rely on stated preferences or proxies, such as damage caused by flooding or the costs of infrastructure to replicate the service provided by soils (e.g. Sustainable Drainage Systems, SuDS) (Bartkowski *et al.,* 2020; Baveye *et. al.,* 2016).

In this report we estimate costs of soil compaction, soil sealing and soil contamination, based on existing costs data sourced from published and grey literature. Searches on Google Scholar, Web of Science and the websites of the Centres of Expertise (ClimateXChange and CREW) and relevant public bodies were searched for literature on valuation of services impacted by soil degradation. Further data were sourced through SEPA and NatureScot. The costs presented here are minimum estimates, based on a limited

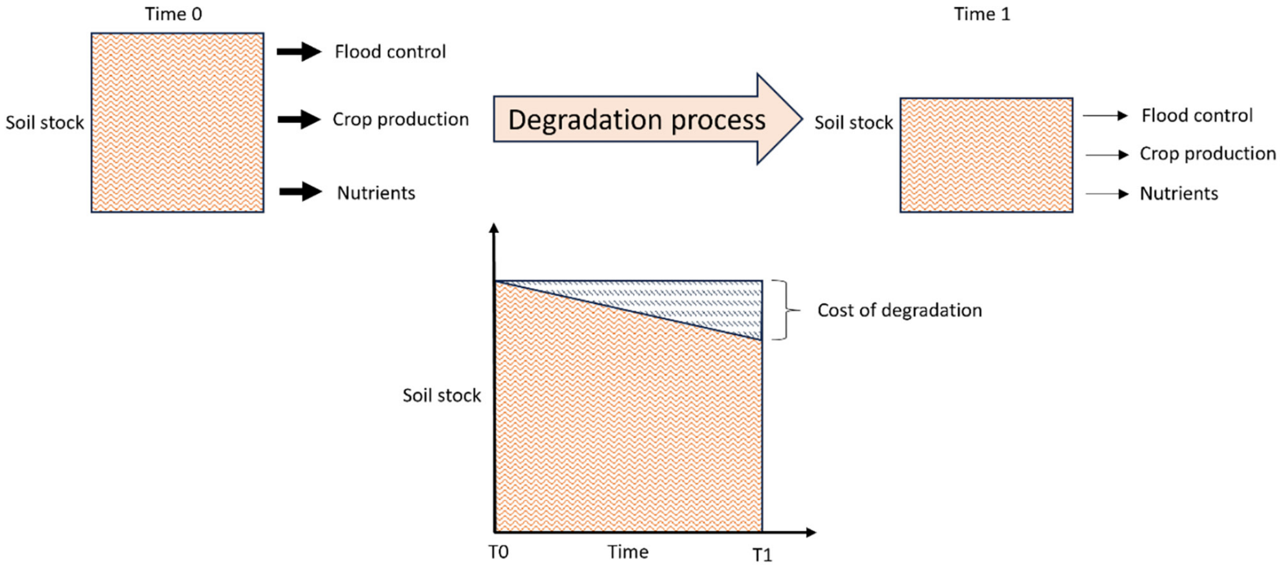


**Figure 1 Change in economic value of soil stocks following degradation.**

| Table 1 Examples of on and off-site costs due to soil degradation processes. | | |
|---|---|---|
| **Degradation process** | **On-site costs** | **Off-site costs** |
| Soil compaction | Yield loss<br>Increased fertiliser needs<br>Increased fuel needs<br>Loss of biodiversity | Declines in downstream water quality<br>Increased flooding<br>Increased GHG emissions |
| Soil sealing | Yield loss<br>Loss of biodiversity | Increased flooding<br>Declines in water quality |
| Soil contamination | Yield loss<br>Contaminated food<br>Loss of access | Declines in downstream water quality |

number of services for which both economic and biophysical data is available. We have only considered market costs, and have not considered preferences for protection, which will likely greatly increase costs of impacts such as flood mitigation.

### 1.3 Structure of the report

We carried out a review of the evidence that is available to both define the extent and cost the impacts of degradation (Figure 2). This includes a detailed assessment of the on and offsite impacts on including flooding, greenhouse gas emissions, runoff and associated sediment loss, nutrient and contaminant losses, crop yield on mineral soils the impacts of degradation on soil biodiversity and an overview of the impacts on peat and peaty soils.

In consultation with the steering group, a stakeholder workshop, and the review of the data available the analysis then focuses on:

- Costing the on and offsite impacts of compaction associated with cultivated land on mineral soils.

Specifically:

- The development of a new method for defining the extent and impact of compaction on soil runoff and water availability.
- Costing the impacts of compaction on crop growth using the candidate crop of Spring barley (The most economically important cereal crop in Scotland).
- The potential impact of the additional runoff on flooding.
- How compaction can be considered alongside the costed impacts of soil erosion on water quality in Rickson *et. al.* 2019.
- Defining the extent and impact of soil sealing on catchment runoff and the potential impact on flooding.
- An assessment of the extent and impacts of contaminants in soils.

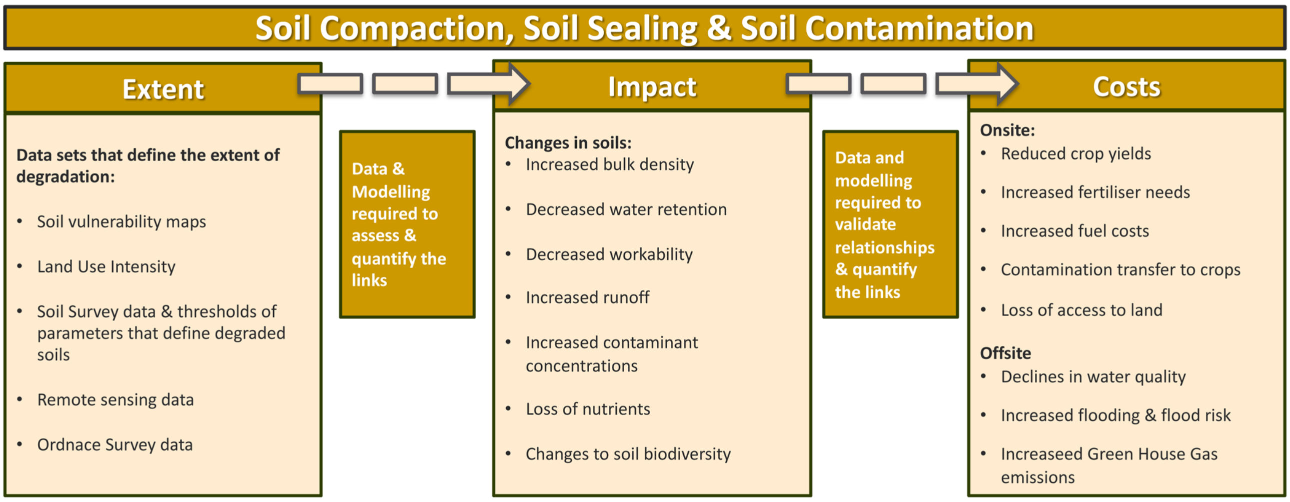


**Figure 2 Framework for assessing the cost of soil compaction, soil sealing and Soil Contamination.**

# 2 Review of impact extent and costs of degradation concepts

Scottish soils, like soils globally, are subject to multiple degradation processes. Degradation processes are those that either reduce the soil stock (e.g., erosion) or impair its ability to provide the ecosystem goods and services required by society. Understanding the impact and extent of these processes is imperative in defining the economic costs associated with them. Impacts of the different processes are, with notable exceptions, relatively well characterised but the extent of degradation is difficult to ascertain at a national level. This review outlines the impacts of soil degradation on soil properties and wider ecosystem services. Costing degradation further requires spatial data where indicators are measured that define whether a soil is degraded and has a sufficient spatial extent to be representative. These data can then be used to support conceptual or process-based modelling approaches to link the degradation to impacts and the severity of those impacts and the data to cost them.

## 2.1 Compaction

**For detailed review of soil compaction see Appendix 1 section 1**

Soil compaction occurs where the soil particles and aggregates are compressed reducing pore volume and increasing soil density. It can also distort soil pores through shearing, so transmission pathways can be disrupted. Compaction can occur under many different land uses either through machinery, livestock or human traffic, or construction of housing or infrastructure (i.e. road, rail, or power networks), however the extent of compaction on specific land uses is not well characterised either on a local or national level. The physical impacts of compaction on soils (decreased pore space and increased density) are generally well understood. With compaction there is less available pore volume to store or conduct water and to house and connect soil biodiversity. Also, plant roots are less able to penetrate the more dense soil to access water and nutrients so crop yields, biodiversity and potentially soil organic carbon content decrease (McGeary *et. al.,* 2022). This can lead to decreased productivity (i.e. decreases in yield or yield quality) and poorer resource use efficiency (i.e. increased time, fuel and fertiliser use for mitigation and cultivation efforts). There are also well-recognised off-site costs from water contamination and greenhouse gas emissions associated with change to flux and process rate through compacted soils. Less appreciated and difficult to calculate indirect off-site costs may be related to changes in soil biodiversity; some organisms will be negatively impacted by damage to their own or their prey species' habitat (e.g. worms and ground feeding birds respectively) while others may be un-affected or even increase. For example, weed species in arable systems may be more abundant due to decreased crop success leaving space and resources for them to exploit. This in turn could have a positive impact on pollinator species.

The amount of land in Scotland affected by soil compaction is still unknown (Lilly *et. al.,* 2018) and attempts to model soil compaction vulnerability, exposure and risk have identified considerable uncertainty (Troldborg *et. al.,* 2013). Maps of soil compaction vulnerability for Scotland indicate that much of the cultivated land is vulnerable to both topsoil and subsoil compaction (Figure 3). While compaction is often associated with narrow, controlled traffic pathways, there is potential for the remaining parts of a field to be trafficked, for example, during harvesting, bailing, silage cutting which may lead to compaction under certain soil conditions. However, although fields may be vulnerable to compaction, this does not imply that they are or will be compacted.

Soil compaction is a major threat to water quality and sustainable crop management. Compacted soils increase runoff with consequences for both flooding and water quality if soil particles entrained within the runoff enter water courses. Compacted soils also limit crop yields due to poor root development restricting access to nutrients and water, in addition to a potential lack of water and poor nutrient cycling in the soil. Where compacted soils become waterlogged for long periods, they are more prone to the release of $N_2O$, a powerful greenhouse gas.

Keller *et. al.* (2019) showed that wheel loads of combine harvesters have grown by approximately 0.14 Mg $yr^{-1}$ (140kg) since the 1960s and that tractor wheel loads have increased by 0.06 Mg $yr^{-1}$ (60kg). In Scotland, vehicles with heavier wheel loads are used in potato harvesting. This increasing size and weight of agricultural machinery means that the soils most at risk of compaction are those

Scotland's soil vulnerability

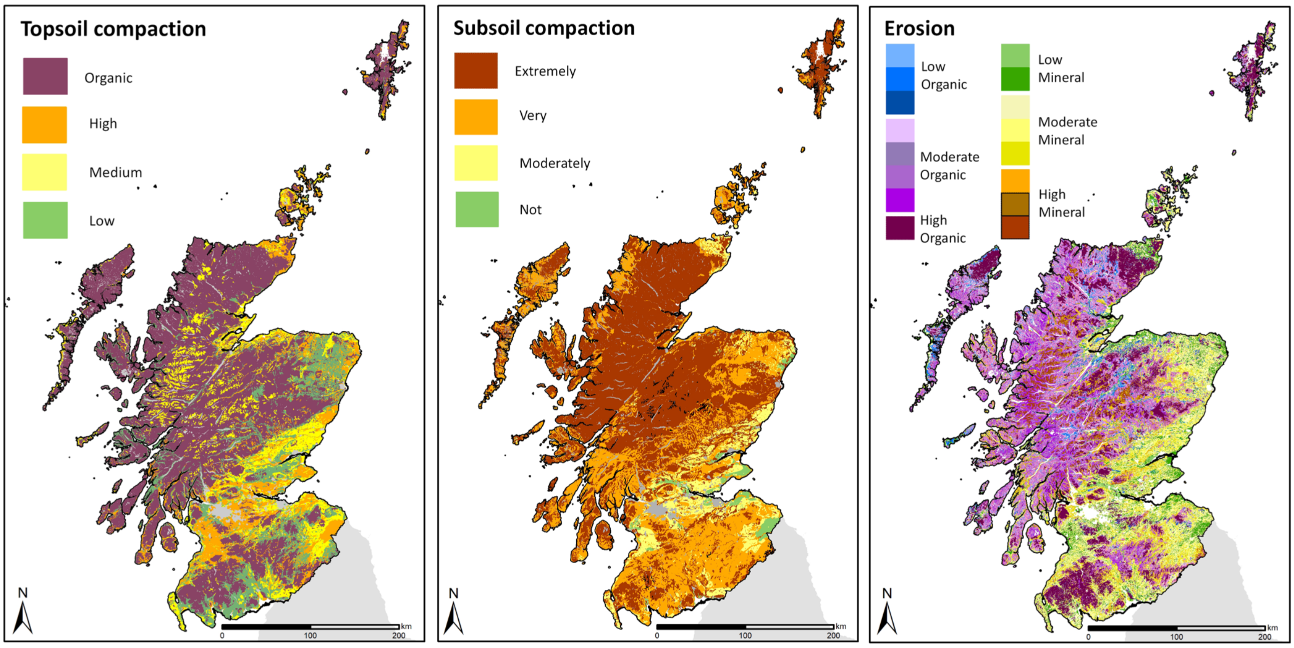




**Figure 3 Topsoil (Left), Subsoil compaction (Centre), Erosion vulnerability (Right) (Baggaley *et. al.,* 2020) from National Soil map of Scotland (Soil Survey of Scotland Staff, 1981) Topsoil compaction: "Organic" :Peaty soils and peat which all have high vulnerability. Erosion: "Low and Moderate organic": Peaty soils "High Organic": Peat.**

under cultivated land, especially where root crops like potatoes are grown. Potatoes are prominent in east Scotland agriculture.

The extent and effects of soil compaction can be exacerbated by more erratic weather due to climate change. Likewise, trends of increased machinery weight have increased stresses on soil (Keller *et al.,* 2019), causing compaction in both topsoil and within deeper subsoil layers that is difficult to ameliorate (Keller and Or, 2022). A compacted soil holds less water and restricts root growth, so productivity under dry conditions can decrease markedly (McGeary *et al.,* 2022) or require more irrigation. A compacted soil is also more difficult to cultivate, so more fuel is used for tillage, and the days available for field operations can be limited if compacted soils drain more poorly.

## 2.2 Soil sealing

Soil sealing can be defined as the destruction or covering of the ground by an impermeable material, which is directly associated with the degree of urbanisation or transport network. Impervious surfaces are mainly artificial structures such as pavements, roads, driveways, car parks and other infrastructure that are covered by impenetrable materials, such as asphalt, concrete, brick, stone, and rooftops (Maucha *et. al.,* 2010).

The European Commission characterises impervious surfaces as "areas in which the soil has been fully, or at least fragmentally, sealed by non-natural materials, leading to the irreversible loss of natural soil functions" (Prokop *et. al.,* 2011).

Soil sealing when it occurs on good quality agricultural land, as well as putting biodiversity at risk, increases the risk of flooding and water scarcity and contributes to global warming (Arnold & Gibbons, 1996, Weng, 2012). In addition, polluted runoff threatens water resources, and the impervious surface area is well correlated to urban heat island affects (Yang *et. al.,* 2020).

A number of methods have been used for mapping soil sealing. These include the use of Landsat (Yang *et. al.,* 2020) and Sentinel satellite images (Copernicus Land Monitoring Service) (Peroni *et. al.,* 2022) and in the UK Ordnance Survey (OS) MasterMap® data. Sentinel satellite images provide higher resolution gridded data (10 m) compared to Landsat images (30 m) (Deliry *et. al.,* 2021). Most studies have focused on the technical comparison of mapping and classification methods (Peroni *et. al.,* 2022) and the machine learning algorithms used to predict the presence of impervious areas (Tobias *et. al.,* 2017; Vaddiraju *et. al.,* 2022). For example, the use of the imperviousness index derived from Sentinel satellite Imagery in Norway showed generally high statistical accuracy, but further analysis showed that there was a 33% underestimation of built-up areas and roads, particularly in rural regions (Strand, 2022). There are, however, limited studies into how the data can be used to quantify the impacts of soil sealing.

## 2.3 Contamination

**For detailed review of soil contamination Appendix 1 section 2**

Soil contamination is regarded as the presence of chemicals or other substances in the soil in concentrations that may be harmful to humans, the environment or biodiversity. Soil contaminants come in many different forms.  This study focusses on xenobiotic (human-made) chemicals which are typically present due to industrial activity, agricultural chemicals, or improper disposal of waste. Six groups are considered (Box 1) as they include contaminants that are well-established through to those that are beginning to emerge as causes for concern. Soil contaminants are not well studied, unlike in waters. Both data on the concentrations of contaminant and the thresholds at which soil functions are impacted are limited. For the more well-established contaminants, there is a reasonable understanding of the mechanisms by which they affect the soil system (the impact), although there is no good understanding of the thresholds at which those impacts occur. Due to the nature of these substances, soil degradation can be highly localised (e.g. heavy metals), while others are more likely to spread in the environment or bioaccumulate in the food chain. This makes understanding their impacts challenging. The impact on biodiversity is likely to be significant but diverse. Heavy metals, for example, are likely to have highly detrimental effects on organisms living in the soil (microbes and mesofauna such as worms and other detritivores) as well as plant life. Pesticides are specifically designed to kill groups of target organisms but are known to have wider effects on non-target species. Pesticide application can lead to an increase in population size when only one pesticide is applied but can lead to reductions and inhibition of basic metabolism when applied in combination.  For other species, such as nematodes, community size and diversity are reduced by the application of any pesticides. The biomagnification of many of these contaminants allows the translocation of contaminants from soil where, in theory, many may be relatively stabilised, through the food chain to cause increased impact elsewhere (Box 1)

**BOX 1: Description of potential contaminants in Scottish Soils and their risks**

**Heavy metals** are widespread in the natural environment. Trace amounts of some metals are essential for the growth and reproduction of plants and animal, but for all metals, once thresholds are exceeded, plant and animal growth is adversely affected. Some metals can accumulate through the food chain, ultimately threatening human and animal health.

**Antimicrobials** play an important role in the prevention and treatment of microbial infections, which has contributed significantly to improvements in human medicine, as well as modern agriculture and animal husbandry. Over-use of antimicrobials may exacerbate natural microbial resistance and promote the spread of resistance genes, reducing the therapeutic potential of antimicrobials for human and animal diseases. Antimicrobial resistance is a global public health crisis and has been increasing through the use and misuse of antimicrobials, which threatens the future effectiveness of infection control.

**Microplastics (MPs)** are non-naturally occurring plastic particles less than 5 µm in size that are widespread in freshwater and terrestrial environments around the world because of their small size and ease of dispersion. These substances are difficult to degrade, which allows them to persist in the environment for hundreds or even thousands of years. They can also accumulate through food chains.

**Persistent organic pollutants (POPs)** are semi-volatile bioaccumulative and toxic synthetic chemicals which are resistant to degradation and can be transported long distances in the environment. A range of substances can be classified as POPs. Some common examples include PCB (Polychlorinated Biphenyls), Polycyclic aromatic hydrocarbons (PAHs) and Pesticides.

**PCB (Polychlorinated Biphenyls)** are widely used in industrial manufacturing, including coolants, capacitors, transformers, fireproofing materials and paints. These pollutants are of serious concern for human and environmental health and have been banned in many countries due to these concerns.

**Polycyclic aromatic hydrocarbons (PAHs)** are toxic POPs that originate from natural and anthropogenic sources and are widespread in the environment. They have properties that mean that they can be transported long distances. 16 PAHs have been identified as priority pollutants by the European Union (EU) due to their potential for bioaccumulation and threat to human health.

Many **Pesticides** are also POPs; although they play an important role in controlling pests and diseases, protecting crops and increasing food production, their adverse effect on biodiversity, ecosystems and public health have also caused concern. The Pesticide Monitoring Program (PMP) was introduced in the United Kingdom (UK) in 1985, and pesticides have been monitored in river water in a number of catchments in the UK since them. However, there are not many studies of the extent of pesticide contamination in terrestrial environment.

## 2.4 Loss of soil carbon

**For detailed review of carbon losses in peat and Peaty soils see Appendix 1 section 3**

Soil organic carbon (SOC) is the carbon component of soil organic matter (SOM), including that held in partially decomposed organisms and byproducts of living organisms (exudates, sloughed cells etc.) at various degrees of physical and chemical availability. Soil organic carbon (SOC) in mineral soils is arguably the most important indicator of soil quality with regard to soil degradation, soil productivity and climate change implications due to its interactions with a myriad of soil functions including soil nutrient cycling, soil water holding capacity, soil structural integrity, crop production, habitat for biodiversity, crop growth, and carbon sequestration (Moinet *et. al.,* 2023). This high degree of connectedness between SOC in mineral (non-peaty) soils and key soil functions shows that SOC loss is likely to be a major indicator of soil degradation in general terms (Guo *et al.,* 2023) but also makes defining the economic cost of SOC loss complex.

Globally SOC stocks are declining (Abdalla *et. al.,* 2020), and many policies and green initiatives promote increasing SOC as part of initiatives to both achieve net zero and mitigate the impacts of climate change. Scotland stores the majority of UK SOC mainly in upland peat and peaty soils (Bradley *et. al.,* 2005; Haygarth & Ritz, 2009), and preservation of SOC in Scottish soils is therefore of high policy importance. Scottish cultivated mineral soils have a median SOC of 3.65% with a stock of 246 ± 9 Mt in cultivated mineral topsoils alone equating to the carbon equivalent of 18 years of Scottish greenhouse gas emissions (based on 2009 levels) (Lilly and Baggaley, 2013). Trends in the carbon stocks in Scottish soils show little evidence of significant change over 10-30 years despite changes in farming practice, climate and other soil fertility markers (Buckingham *et. al.,* 2013; Chapman *et. al.,* 2013; Lilly *et. al.,* 2019; Lilly and Baggaley 2020) and work elsewhere in the UK showing evidence of carbon losses from soils of between 0.6 and 2% per year (Bellamy *et. al.,* 2005).

Peat and peaty soils store 72% of Scotland's topsoil carbon (2049 Mt in top 1m (Rees *et. al.,* 2018)) and are therefore critically important to carbon storage and potential climate feedbacks. Approximately 75 % of all peatland (land with an organic layer > 50 cm thick under semi-natural vegetation) is significantly degraded in Scotland. As a result, carbon is being directly lost to the atmosphere as carbon dioxide and to waterways as dissolved and particulate organic carbon. Much of the carbon is lost to streams and rivers and subsequently lost to the atmosphere but there are high levels of uncertainty as to its fate; how much is metabolised to carbon dioxide and how much is deposited in lakes, rivers, floodplains and the ocean. The RESAS Strategic Research Programme CentrePeat project is working to understand the extent of peatland degradation and the emissions and carbon run-off associated with this degradation. For a review on the impact of erosion, drainage land use conversion of carbon losses from peat soils, see Appendix 1 section 3. In addition to peatlands and peat soils, peaty soils located in uplands and less intensively managed land, store around 800 Mt of carbon and are subject to multiple stressors, including tree planting, fire and overgrazing (see Appendix 1 section 3).

Although there is little evidence of recent significant losses of soil carbon stocks in Scottish soils, this does not mean that Scottish soils are free from the risk of soil carbon loss in the future or have not lost carbon since they started to be cultivated. Given climatic changes to wetter summers and milder winters, soil fauna will be less stressed by temperature and moisture for a larger portion of the year, potentially increasing the break down and mineralisation of soil organic matter. This in theory will be at least partially balanced by increased above ground biomass production but there is still the potential for this to lead to incremental loss of SOC.

# 3 Extent impact and costs of selected degradation processes

## 3.1 Soil Compaction

### 3.1.1 Extent of Soil compaction

Following a review of available data and potential indicators of soil compaction (see Appendix 2 section 1), mineral soils where the proportion of larger pores (>60 µm) was <10% were deemed to be compacted. Pores >60 µm are less able to hold water against the pull of gravity and are the first pores to drain after rainfall allowing air into the soil. The overall volume of these pores is known as the Air Capacity and soils with low Air Capacities, have restricted infiltration (and increased runoff) and remain wetter for longer following rainfall leading to anaerobic conditions which can limit root growth.

A total of 598 sample locations from National Soil Inventory of Scotland (Lilly *et. al.,* 2011), East of Scotland Arable Farm Survey (Valentine *et. al.,* 2012), SoilBio (Loades pers comm) and Glensaugh Farm Grid Survey (Lilly, pers comm) were available to determine the proportion of cultivated topsoils that were compacted (Figure 4) using < 10% Air Capacity as the threshold (see section Appendix 2 section 1). Of these, 158 were at or below this threshold (26 %) which is slightly greater than found by Hallett *et. al.,* (2016) using the Visual Evaluation of Soil Structure (VESS) across four catchments in Scotland. As there were limited data to assess the spatial extent of soil compaction in Scottish soils, the proportion of compacted topsoils in each of the three Topsoil Compaction vulnerability classes was determined to allow spatial extrapolation of the data. The results showed that 56% of soils in the high vulnerability class were deemed compacted, 21% in the moderate vulnerability class and 12% in the low vulnerability class. In the subsequent analysis of crop yields and runoff calculations these proportions are assumed for all cultivated land.

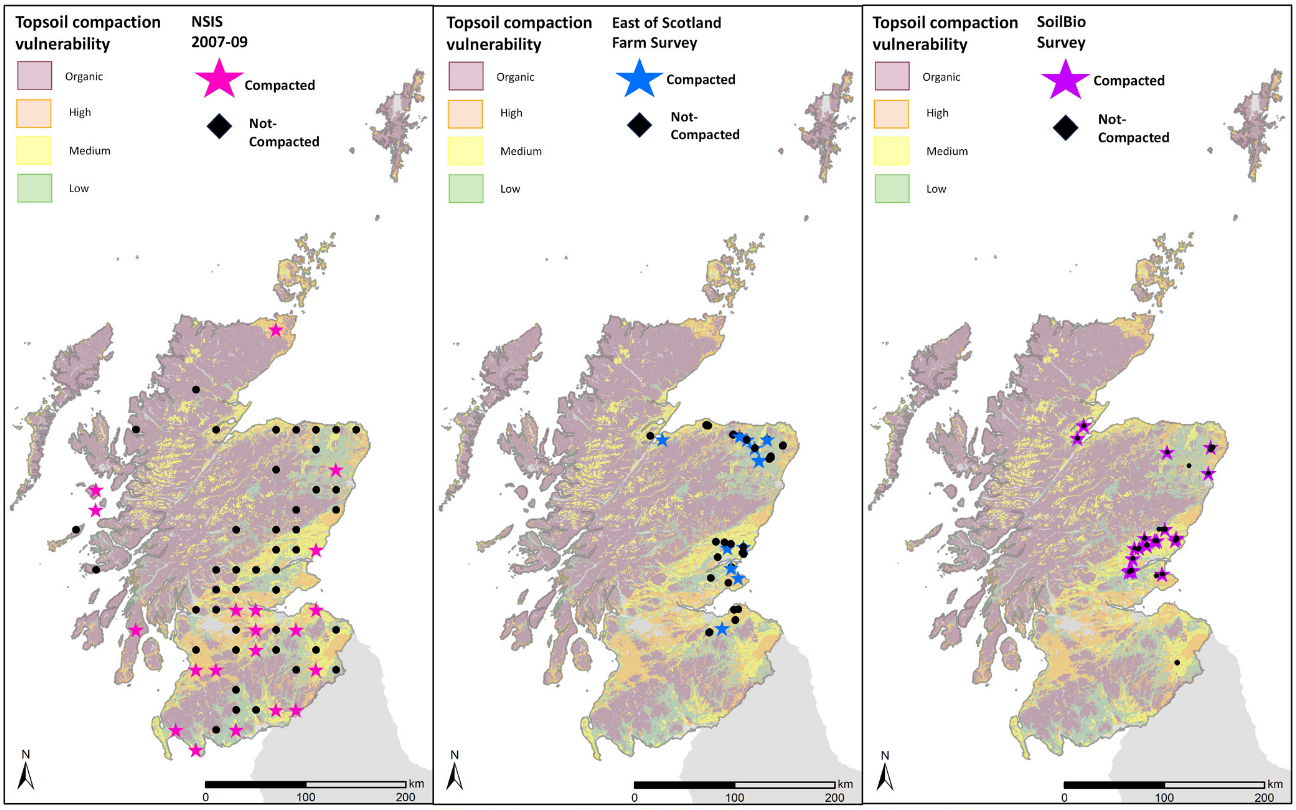




**Figure 4 National data sets used for the assessment of the extent of soil compaction: National Soil Inventory 2007–09, East of Scotland Farm Survey and SoilBio showing compacted points with Ari Capacity less than 10%.**

Most of the above data collections only contain topsoil data and there were not enough data to characterise subsoil compaction in the same way. However, the study by Hallett *et. al.* (2016) using VESS showed that 9% in-field subsoil samples from a mix of arable and grazed land were compacted and that this increased to 20% in areas where vehicle traffic was more concentrated such as those around field gates and turning circles. An attempt was made to relate topsoil VESS scores to soil structure assessments to increase the number and extent of soil compaction assessments. However, there was a poor relationship between bulk density and Air Capacity and this was not considered further.

### 3.1.2 Modelling the impact of soil compaction

Due to the limited amount of measured data at the national scale, it was necessary to adopt a modelling approach to estimate the impact of soil compaction on both surface runoff and on crop yield. The modelling was based on an existing national scale dataset; SSKIB (Scottish Soils Knowledge and Information Base). This dataset comprises sand, silt, clay and organic carbon data for each soil horizon associated with each individual Soil Series (type) delineated on the National Soil Map of Scotland (Soil Survey of Scotland Staff, 1981). Using a set of regression equations (Seismic (GB) pedotransfer functions; (Hollis *et. al.,* 2008; Bradley *et. al.,* 2005; Wösten *et. al.,* 1999), the soil water retention properties and typical bulk density of these soil horizons were derived. As the sand, silt, clay and organic carbon properties of the soil don't change when the soil is compacted, these predicted bulk densities were subsequently used as a means to alter the water retention properties of the soil to model changes in run-off and the impacts on crop growth.

### 3.1.3 Impact of soil compaction on crop growth

Crop growth was assumed to be affected by compaction induced changes in soil physical properties (bulk density and soil porosity), which alter the soil hydraulic and chemical properties of the soil and associated soil water and nutrient flow. The soil compaction impacts of crop growth occur mainly from compaction in the upper layer of the soil (topsoil) but it may also be impacted by compaction in the subsoil.

The impact of soil compaction on crop growth was assessed for a candidate crop (Spring barley) using a Decision Support System for Agrotechnology Transfer (DSSAT) model (Hoogenboom *et. al.,* 2019). Spring barley was chosen as a candidate crop to model as it is the most economically important crop in Scotland.

Firstly, a compacted topsoil bulk density was calculated using the regression equation suggested by Keller and Hakansson (2010) for a soil with specific particle size and organic matter content. As well as a compacted soil bulk density, Keller and Hakansson (2010) suggested that an 'optimum' topsoil bulk density for crop growth would be 87% of the compacted bulk density. These 'compacted' and 'optimum' bulk densities were calculated using the particle size and organic carbon data held within SSKIB for cultivated soils (Figure 5).

A similar approach was used to estimate the bulk densities of a compacted and non-compacted subsoil. Using data published by Nyeki *et. al.* (2017) reference values for compacted subsoils where root growth would be restricted were derived based on soil texture class along with 'Ideal' bulk densities for crop growth (Nyeki *et. al.,* 2017) (Figure 5). These reference values that restrict crop growth are also the values cited in the draft EU Soil Monitoring and Resilience Directive (European Commission, 2023).

These derived bulk densities were then used to estimate the soil hydrological properties for both compacted and optimum topsoils and subsoils using the pedotransfer functions developed for GB soils (Hollis *et. al.,* 2008), and EU soils (Wösten *et. al.,* 1999) for the soils held within the SSKIB dataset. The particle size, horizon thickness and organic carbon contents were held constant and so, only the changes in bulk density were driving changes in soil porosity, water retention and conductivity. The predicted soil hydrological properties for both compacted and optimum soils were then used as inputs to the Decision Support System for Agrotechnology Transfer (DSSAT) model (Hoogenboom *et. al.,* 2019) along with high resolution weather data (1 km) from UK Met Office from 1960 to 2019 (Met Office *et. al.,* 2018) and the 1:250,000 scale soils mapping (National Soil Map of Scotland, Soil Survey of Scotland Staff, 1981). Barley yield was estimated for each unique combination of weather and soil unit for both compacted and uncompacted soils (For full details of the crop modelling see Appendix 2, section 2).

Since soil compaction has a direct impact on the soil hydraulic and chemical properties, two scenarios of yield estimates were considered.

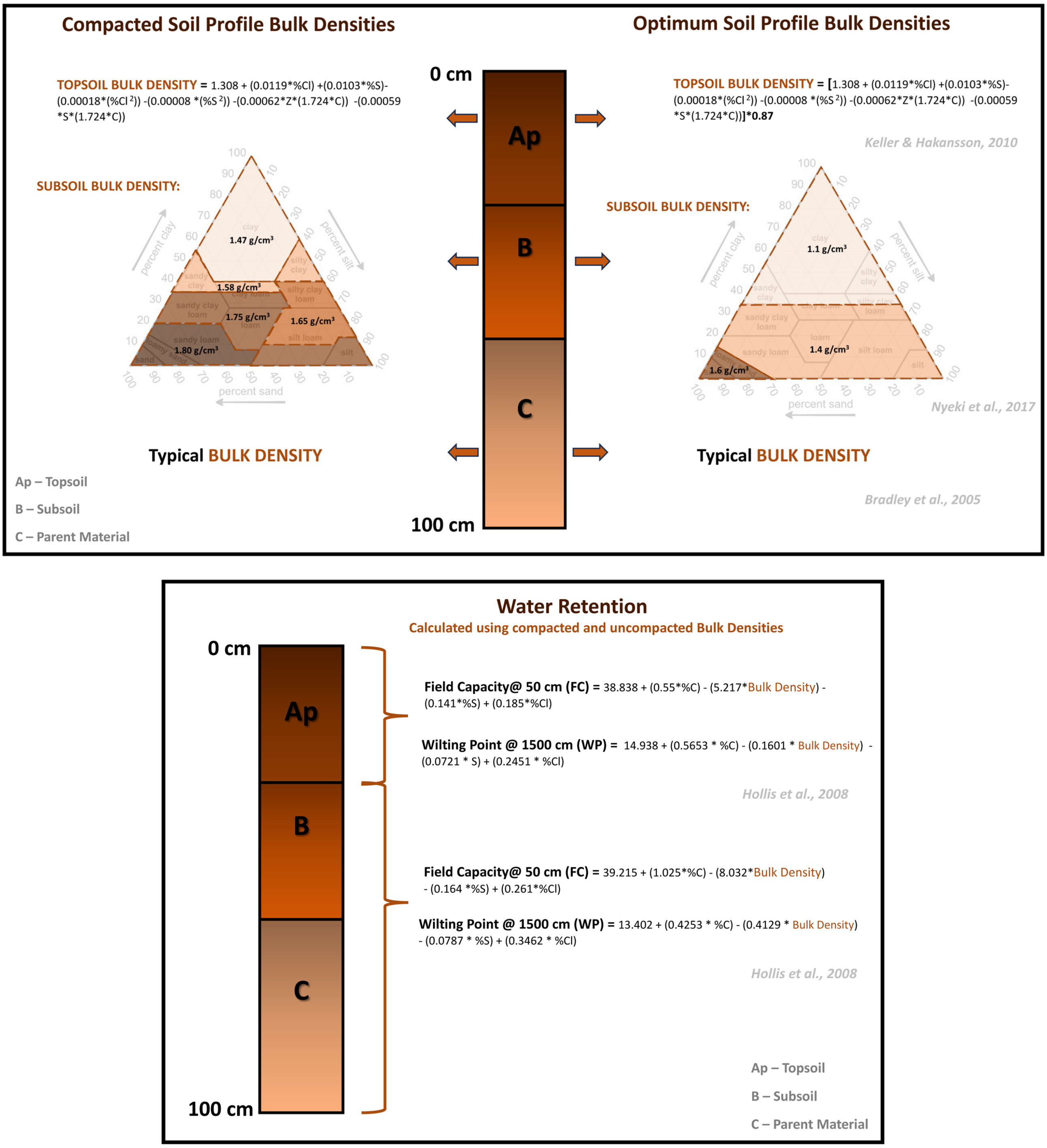


**Figure 5 Bulk density for a compacted and optimum representative soil profile (Ap/B/C horizon sequence) and subsequent modelling of changes in water retention characteristics for input into the DSSAT crop model.**

In the first scenario, the water limited yield potential which is the maximum yield a crop can achieve when only water is a limiting factor was estimated. Any yield decrease would then be attributed to increased drought due to a decrease of the soil water holding capacity and an increase in water loss by drainage and runoff resulting in less water available for crop growth. In the second scenario, water and nitrogen limited yield was estimated. Any additional decrease in yield as compared to scenario one would be attributed to decreased accessibility of nutrients, and increased loss of the soil nutrients by leaching, runoff, and gaseous losses to atmosphere because of soil compaction and/or the interaction of both water and nutrient stress.

The overall impact of soil compaction on barley yield was negative for both scenarios (Figure 6) because of the effect soil compaction had on the soil water holding capacity. Both scenarios included a few cases of no effect depending on the soil hydraulic properties and the amount of precipitation during the cropping season.

The relative yield losses due to soil compaction for the water limited yield ranged between a loss of 18.2 and 0%. These results match with field studies that measured crop yield declines from soil compaction directly (Hallett *et. al.,* 2016). The impact on spring barley yield was the greatest for the soils with a low water holding capacity (less than 160 mm) and with a cumulative precipitation over the cropping season of less than 250 mm. When nitrogen is further considered as a growth limiting factor, the additional yield losses due to nitrogen losses and/or the interaction between both water and nitrogen stresses ranged between 6 and 0%. The additional yield losses were mainly due to increased nitrogen leaching and a slight increase in nitrous oxide emissions, resulting in a decrease in nitrogen availability to crop growth. This led to a decrease in nitrogen uptake which had a direct impact on grain nitrogen content and thus grain quality.

The crop model selected for this study does not consider, the mechanical impedance of the soil to root growth. Valentine *et. al.* (2012) found that mechanical impedance explained a reduction in root growth rate using a seedling assay with intact soil cores taken from across the east of Scotland.

With additional project resources, a future project could consider mechanical impedance effects in greater detail, as they may exacerbate the yield impacts that we observed with the modelling approach deployed.

### 3.1.4 Impact of soil compaction on additional runoff

Air Capacity thresholds which are used to determine if a soil is compacted, can also be used to determine the additional runoff (and, hence, increased flood risk) when a soil is compacted. Similar to the approach used to explore the impact of soil compaction on crop yields the Hollis *et. al.* (2008) pedotransfer functions (see section 3.2.1, Figure 7) were used to predict the water retention properties of Scottish soils delineated on the 1:250 000 scale National Soil map of Scotland (Soil Survey of Scotland Staff, 1981). In this case these were based on the uncompacted soil profiled using typical bulk densities from Bradley *et. al.,* (2005), and compacted soil profiles using the Keller and Hakansson, 2010 bulk densities for topsoils and the Nyeki *et. al.,* (2017) bulk densities for subsoils (Figure 7).

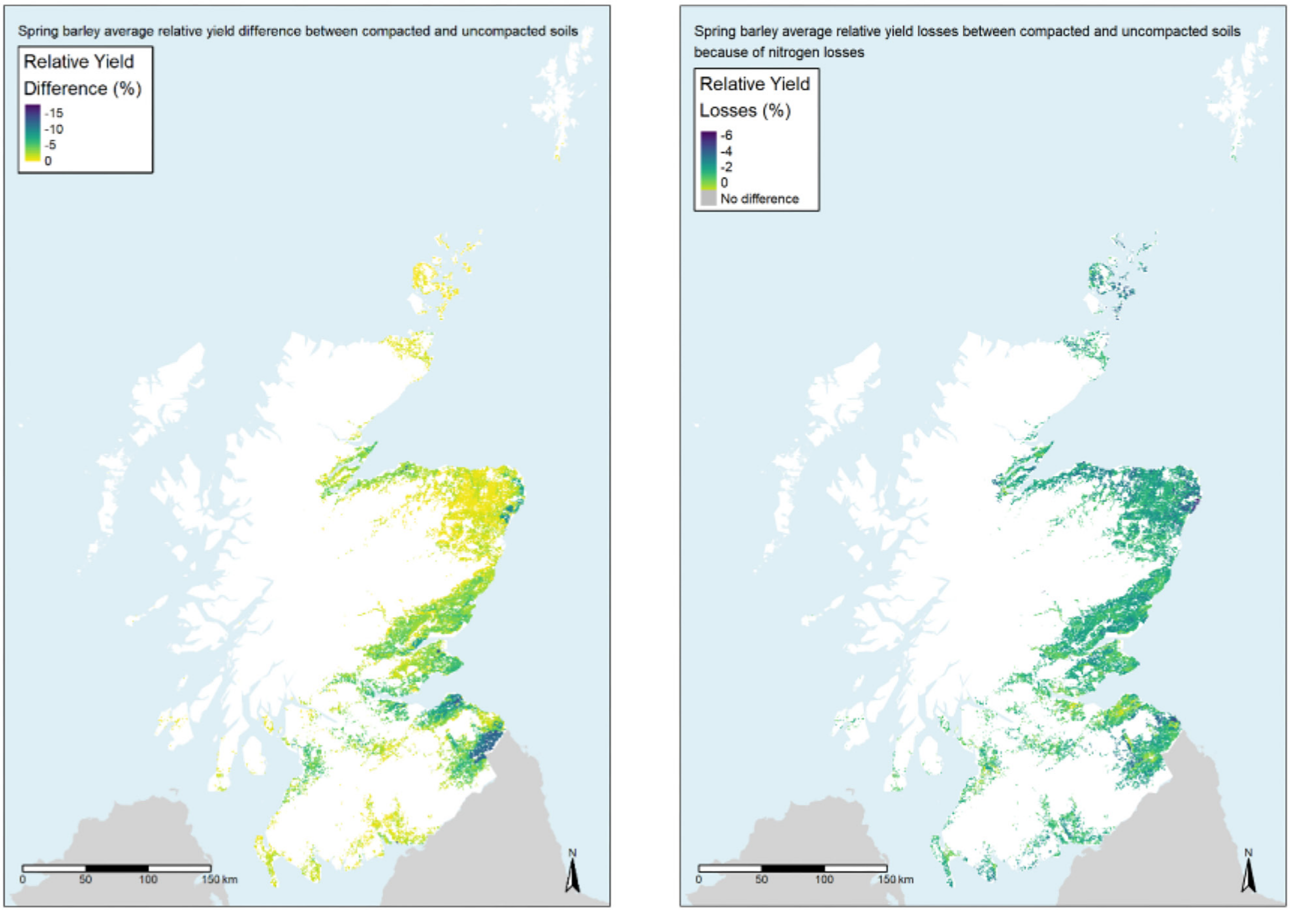


**Figure 6 Average spring barley relative yield losses because of soil compaction for the water limited yield (left) and the additional relative yield losses due to nitrogen stress (right). Assuming all soils are either compacted or not compacted.**

When calculating changes in Air Capacity of compacted soils, the whole topsoil was assumed to be compacted along with the upper 15 cm of the subsoil (or the whole of the second horizon, which ever was greater), effectively simulating the presence of a plough pan (Figure 7). Such an assumption is based on the recognised attenuation of surface soil stresses through depth in the soil, confirmed in many measurements of bulk density or visual assessment of soil compaction with depth. One exception to this were the soils developed on glacial lodgement till and have a naturally compact and slowly permeable subsoil. For these soils, we assumed that the subsoil could not be compacted any further (Jones, *et. al.,* 2003).

The loss of large pores in compacted soils means that water holding capacity is reduced, and run-off is likely to increase. Using the pedotransfer functions, the difference in percentage of Air Capacity lost between compacted and typical soils for topsoil and the upper part of the subsoil was calculated. This provides an estimate of the loss of large pores that allow water to infiltrate and provide storage for rainfall events and, hence,

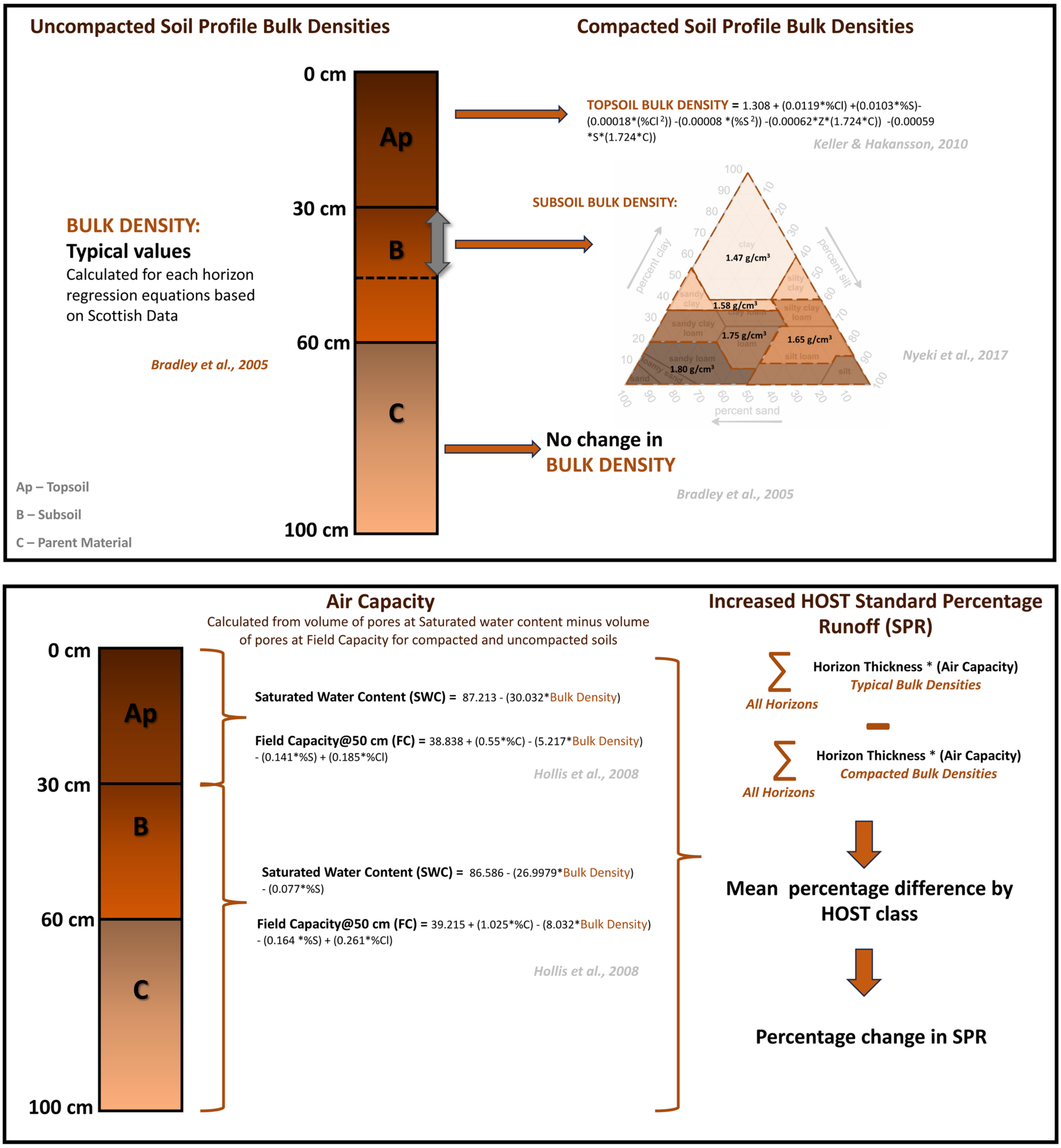


**Figure 7 Bulk density for a compacted and uncompacted representative soil profile (Ap/B/C horizon sequence) and subsequent modelling of Air Capacity and increases in Standard percentage runoff by HOST class for compacted soils which was then be used to model changes in catchment SPR values.**

the likely additional runoff. Where the differences were negative, that is, where the water retention predicted using the 'typical' bulk densities were greater than that predicted using Keller and Hakannson (2010) or by Nyeki *et. al.* (2017), these were reset to zero, suggesting no change due to compaction. This was a pragmatic decision to allow average change in Air Capacities by the Hydrology of Soil Types (HOST) classes (Boorman *et. al.,* 1995) to be calculated. HOST classifies UK soils based on their ability to transmit water both vertically and horizontally. It has been used to derive hydrological indices such as Standard Percentage Runoff (SPR).

The average percentage changes in Air Capacity were then used to adjust the Standard Percentage Runoff (SPR) derived from the HOST classification (Boorman *et. al.,* 1995) for cultivated soils. For example, if the HOST SPR was 20% and the percentage reduction in Air Capacities was 10%, the revised SPR was increased by 10% to 22% suggesting an additional runoff of 2% of any rainfall event. This approach is similar to that used by Holman *et al.* (2003) but where they used arbitrary increases in SPR, we have used the changes in Air Capacity to guide these changes in relation to specific HOST classes and soil properties.

HOST SPR is used to calculate the proportion of runoff relative to rainfall from a catchment by multiplying the HOST class SPR by the area of each HOST class within a catchment. Comparing both

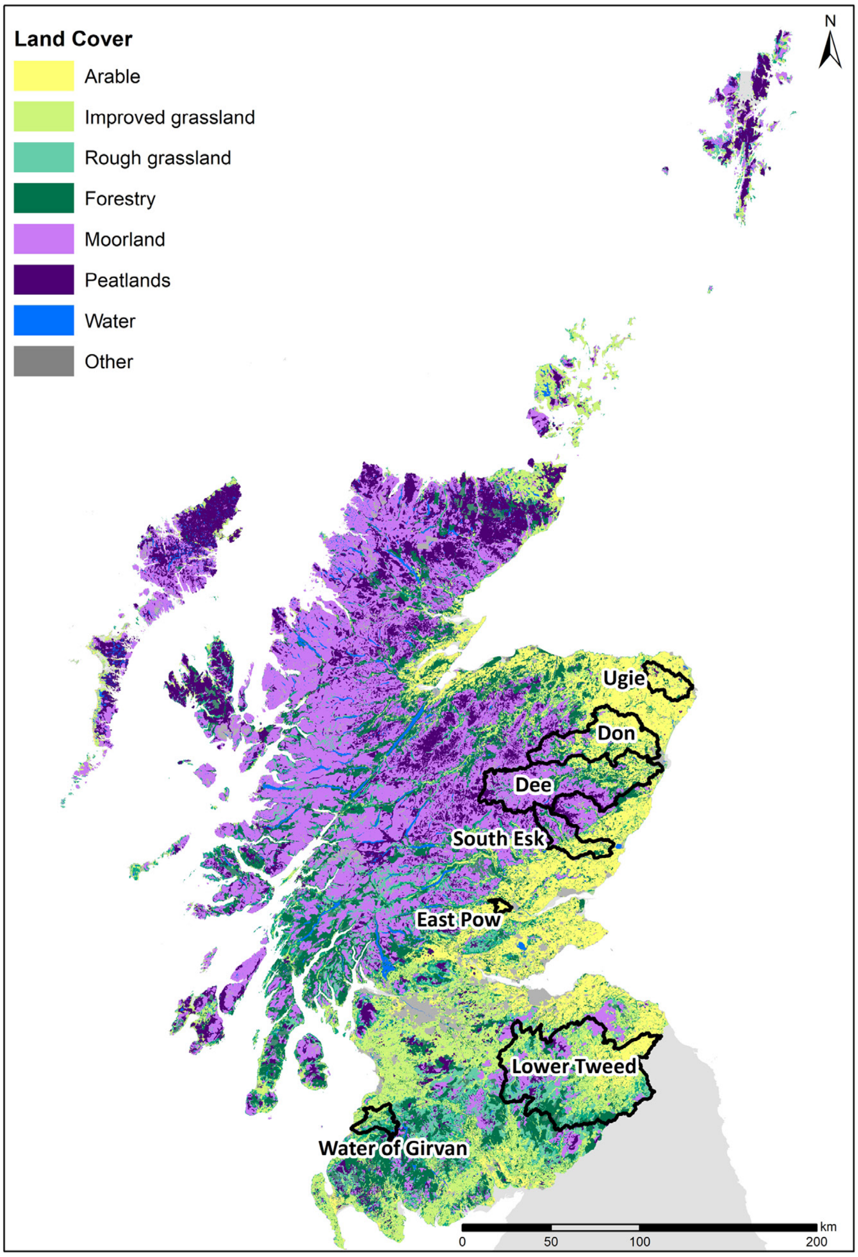



**Figure 8 Seven catchments where additional runoff due to compaction is modelled. Broad Land cover classes cover from LandCover Map of Scotland 1988 (LCS88). (Macaulay Land Use Research Institute, 1993).**

the SPR values given in Boorman *et. al.* (1995) with the adjusted SPR allows an assessment to the potential additional runoff due to soil compaction to be made. To fully assess the impact of the increased runoff on flood extents requires the implementation of a hydrological model to determine the actual flow hydrograph combined with flood susceptibility and flood plain characteristics and was out with the scope of this project. However, the work done to use HOST SPR to calculate increased runoff for compacted soils could be fed into the SEPA flood modelling approach which uses the Flood Estimation Handbook software and HOST runoff grids as inputs.

The additional SPR was calculated for seven catchments (Figure 8) ranging in size from 48 to 3378 km$^2$ and with differing levels of cultivated, moorland and forestry and different soil types. These are the five case study catchments selected as part of the project to cost the impacts of soil erosion (Rickson *et. al.,* 2019) and the two catchments (The Dee and the Don) where the impacts of soil sealing have also been assessed as part of this project (Section 3.2).

We estimate a lower range, where the proportion of soil compacted is based on topsoil vulnerability to compaction estimated (Section 3.1.1: 56% of soils in the High vulnerability class deemed compacted, 21% in the Moderate vulnerability class and 12% in the Low vulnerability class), and a higher range, based on 100% compaction of cultivated soils. The changes ranged from an increase in SPR of 0.2% to a change of 2% for the lower compaction level and from 1.2% to 7.8% assuming all cultivated land in the catchment was compacted. The greatest percentage changes are seen in the Ugie and Pow which have the greatest proportion of cultivated land (Table 2).

The Pow catchment was used to show that the runoff vulnerability class (Lilly and Baggaley, 2018) increases when soil profiles become compacted (Figure 9). Since the runoff vulnerability is a component of the erosion risk model (Lilly *et. al.,* 2002), it suggests that there could be additional nutrient and sediment export from compacted soils. In Rickson *et. al.* (2019) erosion estimates of sediment yield and nutrient losses took account of land use intensity and land management practices that can lead to soil compaction, so to some extent their estimates include a soil compaction impact.

Since the calculated losses due to erosion (Rickson *et. al.,* 2019) contain a proxy for other soil degradation processes it remains difficult to estimate the potential loss of nutrients and the impact on water quality from eroded sediment and associated nutrients such as phosphorus from soil compaction alone. The exports in Rickson *et. al.* 2019 are therefore the best estimates of nutrient exports to water courses from soil degradation and further work is needed to unpick the interrelationships and generate a new set of rules and models that allows the calculation of the combined impact of compaction and erosion on nutrient and sediment exports to waters while accounting for the interactions.

**Table 2 Catchment runoff under 2 compaction scenarios: SPR1: proportion of cultivated land compacted SPR2: all cultivated land is compacted; Calculations based on compaction extent and compaction risk.**

| CATCHMENT | SPR without soil compaction | Catchment area (km$^2$) | Percentage of cultivated land in the catchment | SPR 1 – A proportion of cultivated land by vulnerability class is compacted | Percent change to SPR1 | SPR2 – All Cultivated land is compacted | Percent change to SPR2 |
|---|---|---|---|---|---|---|---|
| South Esk | 40.6 | 563 | 39 | 40.9 | 0.8 | 42.2 | 4.1 |
| Lower Tweed | 36.8 | 3378 | 45 | 37.2 | 0.8 | 37.9 | 2.9 |
| Dee | 38.9 | 2077 | 21 | 39.0 | 0.2 | 39.4 | 1.2 |
| Ugie | 33.7 | 333 | 84 | 34.3 | 1.7 | 35.6 | 5.6 |
| Pow | 38.1 | 48 | 79 | 38.8 | 2.0 | 41.1 | 7.8 |
| Girvan | 42.2 | 251 | 37 | 42.5 | 0.7 | 43.1 | 2.2 |
| Don | 32.0 | 1312 | 56 | 32.3 | 0.9 | 33.3 | 4.0 |

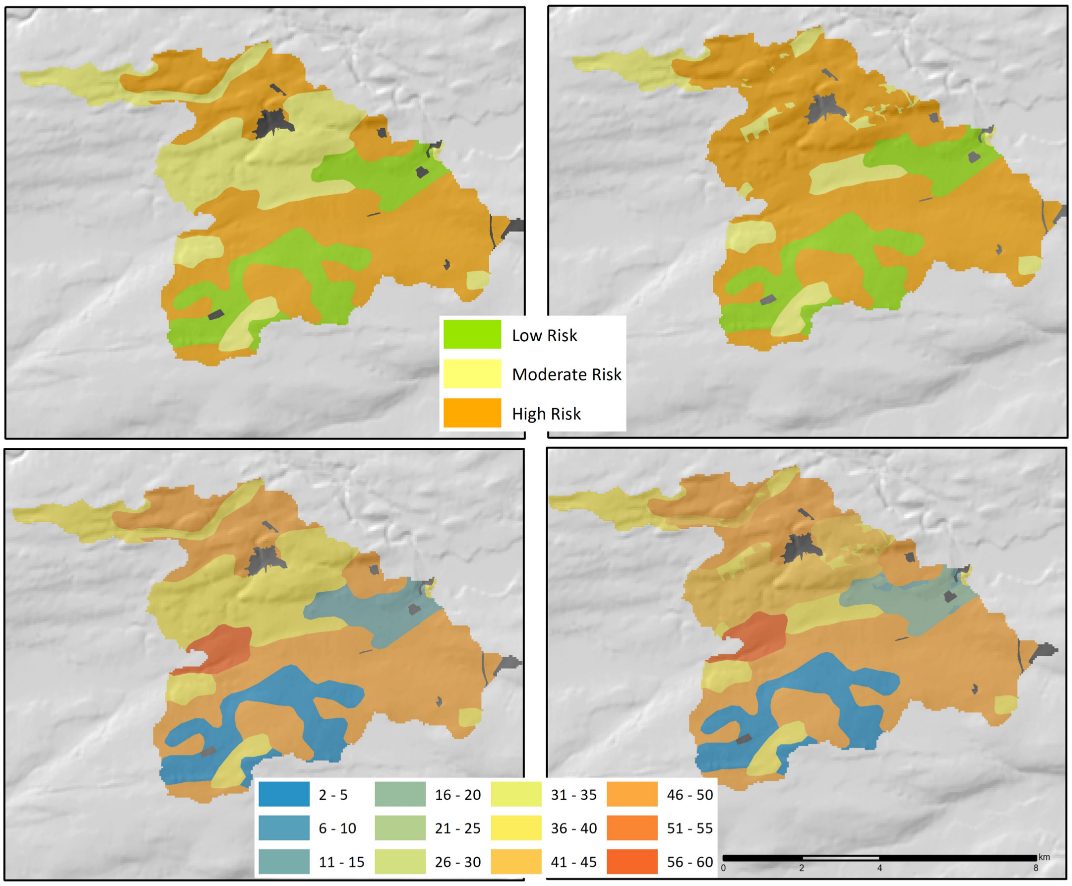




**Figure 9: River Pow catchment with HOST Standard Percentage Runoff for cultivated soils with: Left: Typical bulk density profile (Figure 7); Right: Compacted bulk density profile (Figure 7); Top: SPR runoff values; Bottom: Runoff Vulnerability Class**

### 3.1.5 Costs of Compaction

Costs of flooding to property because of soil compaction can be estimated from the insurance claims following flooding events. In two Scotland-wide surveys a per property cost of each flood event was estimated to be £56,776 – £76,035 (Owusu *et. al.,* 2015; Werritty, 2007). Estimating the total cost to Scotland would require modelling impacts at the property level (Table 3), which is beyond the scope of the current modelling (Section 3.2.2).

The average annual damage of flooding to local authorities is available from Flood Risk Management Plans (SEPA, 2021), estimated at £259 million/year. An increase in flood events or severity of flooding would therefore see a potential increase in costs, with an estimated £2.6 million per percentage increase in flood events or flood severity. Further work is needed to link the increases in catchment Standard Percentage Runoff to flood extent and severity. In addition, care must be taken with this figure however, as the location of increased flooding will impact damage inflicted, and it does not take into account any grey or green infrastructure implemented to reduce damages.

Soil compaction is also associated with a loss of agricultural yield. Using the published farm prices per tonne of barley for 2023 we estimate the cost of reduced yield due to compaction to range from £16,421,893 based on estimated current levels of compaction to £48,632,483 assuming all arable land within Scotland is compacted (Table 4).

**Table 3 House insurance claims following flooding event from Owusu *et al.,* 2015 and Werritty, 2007.**

| Cost type | Cost | Cost per property | Number of properties flooded | Total costs to Scotland |
|---|---|---|---|---|
| Insured building costs | £39,553 - £51,000 | | | |
| Insured contents costs | £13,778 - £21,600 | £56,776 - £76,035 | Unknown | Unknown |
| Uninsured costs | £3,435 | | | |

| Table 4 Crop losses due to soil compaction. | | | | | |
|---|---|---|---|---|---|
| **Catchment** | **Yield Loss (t)**<br>**(100% compacted)** | **Yield Loss (t)**<br>**(Compaction areas as calculated)** | **Average barley price/t** | **Total Loss**<br>**(100% compacted)** | **Total Loss**<br>**(Compaction areas as calculated)** |
| Dee | 5871 | 1521 | £150 | £880,623 | £228,098 |
| Don | 10648 | 3220 | | £1,597,175 | £483,019 |
| Girvan | 1079 | 438 | | £161,904 | £65,682 |
| Lower Tweed | 40084 | 14031 | | £6,012,596 | £2,104,603 |
| Pow | 1384 | 335 | | £207,662 | £50,279 |
| South Esk | 5770 | 1008 | | £865,544 | £151,272 |
| Ugie | 5539 | 2399 | | £830,904 | £359,905 |
| | | | **Total loss across all measured catchments** | **£10,556,408** | **£3,442,857** |
| | | | **Scotland wide** | **£48,632,483** | **£16,421,893** |

An additional compaction cost related to agriculture is related to increased fuel use required for additional or more complex field operations. Scaling up the increased fuel use per hectare we estimate 29-87%, dependent on soil type (Graves *et. al.,* 2015), additional fuel use as a result of soil compaction. With non-compacted fuel use estimated at 60 to 270 litres $ha^{-1}$ $yr^{-1}$ dependant on crop type, with the lower end being associated with cereals and the upper with growing potatoes (Roberts *et. al.,* 2023). Assuming an increase in fuel use based on soil texture (Table 5) we estimate an additional 17.4–234.9 litres $ha^{-1}$ $yr^{-1}$, with lowest increase based on sandy or peaty topsoils, and highest increase for clay topsoils soils (Chamen *et. al.,* 2017, Graves *et. al.,* 2012)  Based on the average cost of red diesel for 2023 (£0.89/l), this has an estimated cost of £15–£209 $ha^{-1}$ $yr^{-1}$, with national additional fuel use on arable land of between £9,000,000 increasing to £26,000,000 if all arable soils are compacted.

As discussed in section 3.1.3 the losses of eroded nutrients and sediments are likely to increase if compaction is considered. When considering erosion in isolation previous studies have estimated the cost of nutrient replacement as £10.1million (Rickson *et. al.,* 2019). We would expect compaction to increase this due to the resulting increases in erosion. Costs of fertiliser have also increased more than 3 fold since 2019 (Table 6), which will impact farmer behaviour in fertiliser application. Therefore, although we are confident that nutrient replacement costs will increase, without modelling the additional losses due to compaction we cannot estimate these costs. Forthcoming work in the CREW project on “Mitigating the impacts of climate change on the water quality of standing waters” (May *et. al.,* 2024) will provide updated costs of removing (in-lake) phosphorus, which will further improve these values.

| Table 5 Increase in fuel use on compacted soils. Proportion of compacted land. | | | | | |
|---|---|---|---|---|---|
| **Crops** | **Fuel Use** | **Fuel Costs** | **Sandy topsoils** | **Silty topsoils** | **Clay topsoils** |
| | | | **Additional cost (£/Ha)** | | |
| | **(l/Ha)** | | **(Increase - 29% )** | **(Increase - 60%)** | **(Increase - 87%)** |
| **Potatoes** | 270 | £0.89/l | £70 | £144 | £209 |
| **Cereals** | 60-72 | | £15-£19 | £32-38 | £46-£56 |
| **Oil Seed Rape** | 77 | | £20 | £41 | £60 |
| **Peas & Beans** | 71 | | £18 | £38 | £55 |
| **Area of arable land** | **8692 km²** | | **Additional 100% compacted** | | **£26,000,000** |
| | | | **Additional compaction area as calculated** | | **£9,000,000** |

| Table 6 Updated per unit costs of erosion associated with nutrient losses and water quality impacts. | | |
|---|---|---|
| **Impact** | **Cost estimated in Rickson *et al.*** | **Updated cost** |
| **On-site costs** | | |
| Fertiliser (N) | £670/t | £2,430/t (SAC 2022) |
| Fertiliser (P) | £680/t | £2,010/t (SAC 2022) |
| Fertiliser (K) | £450/t | £1,280/t (SAC 2022) |
| **Off-site costs** | | |
| Greenhouse gas emissions | £68/t | No updated costs available. |
| Environmental water quality (nitrate in rivers and canals) | £190/t | |
| Environmental water quality (nitrate in transitional waters) | £10/t | |
| Environmental water quality (phosphorus in freshwater lakes) | £1,407/t | |
| Drinking water quality (nitrate) | £203/t | |
| Drinking water quality (sediment) | £18/t soil | |

## 3.2 Extent of Soil Sealing

### 3.2.1 Extent of Sealing

Two data sets have been used to estimate the extent of soil sealing for Scotland in the Dee and Don catchments. These catchments were chosen as representative of recent expansion of both infrastructure and settlement onto cultivated and semi-natural land due to the construction of the Aberdeen Western Peripheral Road and associated increase in extent of residential settlements.

The first dataset used was the Copernicus Land Monitoring products which provide imperviousness density maps derived from sentinel satellite data which provides a gridded data set with an imperviousness index for land ranging from 0–100. The second was the Ordnance Survey (OS) MasterMap® Topography Layer for the United Kingdom which is obtained from areal photogrammetry with high spatial (1m) and temporal resolution (updated every 6 weeks). (Figure 10).

For the Dee and the Don catchments the OS MasterMap® data provides similar sealed extents to the Copernicus data however it gives more precision and accuracy with much lower uncertainty compared to Copernicus satellite data layers. For example the OS MasterMap® data provides a more accurate identification of the extent of small and linear features such as individual properties and roads and tracks (Figure 10). A further benefit of the OS data are that it provides a detailed classification of both sealed and unsealed land parcels. For example buildings can be distinguished from other areas of sealed land and agricultural land distinguished from different types of forestry and moorland.

OS MasterMap® data have been successfully used by NatureScot for evaluation of the extent of soil sealing which is reported as a Scottish national resilience indicator. In this data set artificially sealed areas are defined by the "Make" category being equal to "Manmade" and the "Desciption Group" category not including the term "Landform".

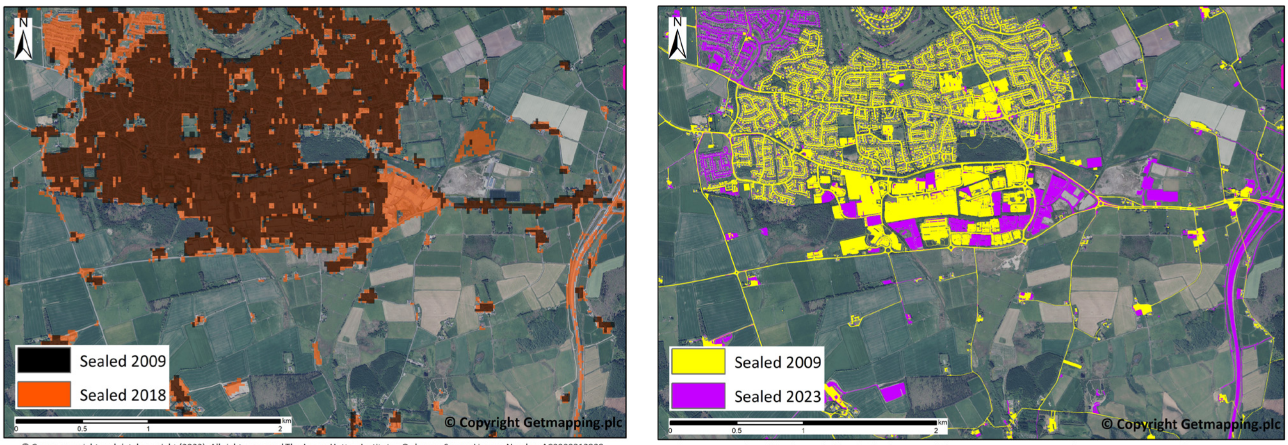


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**Figure 10 (LHS) Imperviousness Density Layer; European Union's Copernicus Land Monitoring Service information. https://land.copernicus.eu (RHS) Ordnance Survey MasterMap® Topography Data Showing development of Aberdeen Western Peripheral Road and growth in Westhill. Base Map: Ordance Survey 1:50,000 Raster data.**

(NatureScot: https://www.environment.gov.scot/our-environment/state-of-the-environment-previous-reports/ecosystem-health-indicators/resilience-indicators/indicator-13-soil-sealing/). Hence, we have used the OS MasterMap® with the same classification of sealed areas for the evaluation of the change of the extent of sealing within the last 14 years (2009–2023) and its potential impact on runoff.

The whole Don and Dee river catchments cover an area of 3407 km$^2$, with Dee catchment area being 2086 km$^2$ and Don catchment area 1321 km$^2$. The OS MasterMap® data showed that 21.5 km$^2$ and 27.5 km$^2$ of land are artificially sealed in the Dee and Don catchments, respectively, in 2009. In 2023, the sealing extent across Dee and Don catchments has increased to 28.5 km$^2$ and 36.8 km$^2$, respectively. The results indicate that soil sealing in the whole Don and Dee catchments has increased by 20 % in the last 14 years. This sealing has mostly occurred in or around urban and suburban areas with the evident emergence of the Aberdeen Western Peripheral Road (Figure 10)

### 3.2.2 Impact of sealing on runoff

In order to assess the impact of the increased extent of soil sealing in the studied catchments, the Hydrology of Soil Types (HOST) (Boorman *et. al.*, 1995) classification was used. Predicted runoff for the catchments was generated using the I:250 000 scale National Soil map of Scotland (Soil Survey of Scotland Staff, 1981) combined with the sealed areas from the OS MasterMap® data. The area weighted standard percentage runoff (SPR) value based on the HOST class of each of the soils, and an SPR value of 70% (Environment Agency, 2022) for sealed areas identified from the OS data from 2009 was calculated (Figure 11). Where land had subsequently become sealed by 2023 the SPR from the underlying soil was updated to 70% to represent the additional runoff from the catchment and the catchment weighted SPR values were updated. The differences in the catchment weighted SPR for 2009 and 2023 is an estimate of the differences in runoff and potential increased flood risk associated with the additional sealing.

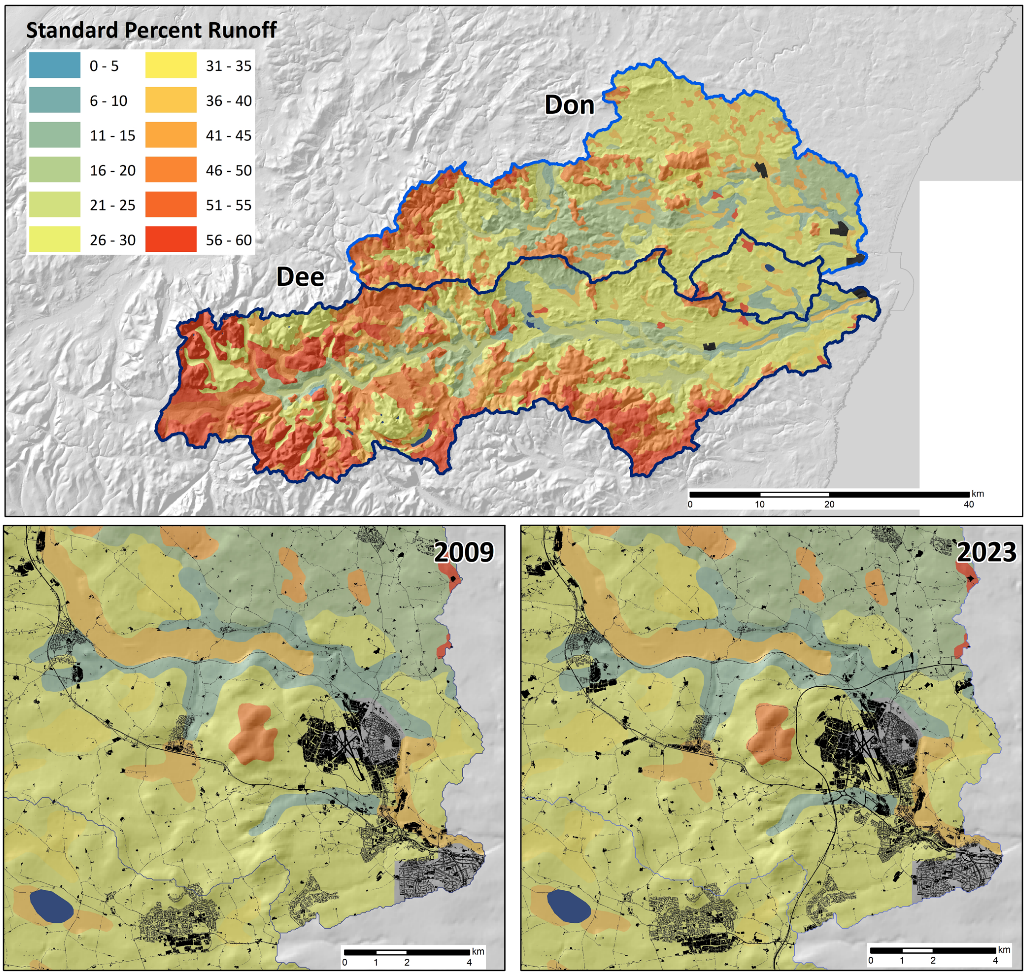




**Figure 11 Standard Percentage Runoff (SPR) values from HOST for the whole Dee and the Don catchments and in sealed area from OS MasterMap® data extracts for 2009 and 2023 around new Aberdeen Western Peripheral Road showing increased in sealed areas (black area on map).**

**Table 7 Summary calculations of sealed areas and their effect on increased runoff percentage in Don and Dee river catchments in 2009 and 2023.**

| Catchment | Area ($km^2$) | Sealed area ($km^2$) in 2009 | Sealed area ($km^2$) in 2023 | SPR for sealed areas (%) | Weighted SPR for the catchment In 2009 | Weighted SPR for the catchment In 2023 |
|---|---|---|---|---|---|---|
| Dee | 2086 | 21.5 | 28.5 | 70 | 38.89 | 39.03 |
| Don | 1321 | 27.5 | 36.8 | 70 | 32.26 | 32.54 |
| Westhill sub-catchment of the Dee | 149 | 4.38 | 6.11 | 70 | 31.51 | 31.99 |

The distribution of sealing across the Don catchment has led to a greater increase (0.9%) in the total runoff compared to the Dee catchment increase (0.4%) in 2023 compared to 2009 (Table 7). The sealed area in the smaller Westhill sub catchment has increased from 3% to 4.1% and resulted in an increase of 1.5% in SPR. This additional runoff is a similar order of magnitude to the additional runoff estimated as a result of soil compaction. To fully assess the impact of the increased runoff on flood extents requires the implementation of a hydrological model. As with compaction the use of HOST SPR to calculate increased runoff for sealed soils could be fed into the SEPA flood modelling approach which uses the Flood Estimation Handbook software and HOST runoff grids as inputs.

### 3.2.3 Costs of sealing

To evaluate the costs of soil sealing we have estimated the costs associated with increased flood events using the same approach as that of soil compaction (Table 3). In addition to flood damage downstream of sealed areas there is the additional risk of damage to property where sealing occurs in areas already at risk of flooding. However, as with compaction, we are not yet able to model the number of properties expected to be impacted, and so cannot estimate the full Scottish costs.

Local authority costs of flood damages can also be used to estimate soil sealing costs compared to the £259 million currently estimated (SEPA, 2021). If we assume a linear increase in costs with increased flooding, a 1% increase in flooding would equate to a cost of £2.6 million in local authority costs. As with the compaction costs it must also be recognised that cost will vary spatially, and we have not accounted for improvements in flood protection.

The cost of mitigation measures set in place to reduce flooding as a result of soil sealing also represents a cost, although this cannot be added to the damage costs reported above. Sustainable Urban Drainage Systems (SUDS) in Dunfermline ranged in cost from £69,000 to £273,000 over a 50-year lifespan (Wolf *et. al.,* 2015). The particular characteristics of each SUDS installation will mean that costs will vary across Scotland, and further research would need to be conducted to estimate the full costs of SUDS installed.

In addition to property, infrastructure, and Local Authority costs there may be significant private costs of flooding, that we have not accounted for. These may include loss of crops, access to recreation areas, and impacts on mental and physical health (Currie *et. al.,* 2020). The sealing of soil also takes it out of agricultural production, leading to a cost in yield loss. However, because the change in land use brings other societal value, such as transport links, housing, and amenities, estimating the societal cost of the loss of agricultural land is beyond the scope of this work.

## 3.3 Soil contamination

### 3.3.1 Extent of contamination

There is limited information available in Scotland to assess the overall extent of soil contamination against set thresholds. Thresholds for contaminants can vary depending on the elements or compounds considered. Many of the existing thresholds refer to the amount of contaminant that might cause the soil health problem and would help to protect the human health (Table 8). There are some soil quality standards from USEPA listed in the table below although this is not a complete list (some might only contain part compounds of the group of chemicals).

A national picture of concentrations of potential contaminants in soils is available from National Soil Inventory of Scotland (NSIS2007–2009) where most of the contaminant concentration ranges were within the thresholds set by the USEPA's soil quality standard. Once added to soil, most metals are strongly retained and losses through erosion

| Table 8 Types of contaminants and data where they have been measured in Scotland. | | | |
|---|---|---|---|
| **Contaminant** | **Sources** | **Thresholds** | **Extent** |
| Heavy metals | Natural (e.g. rock weathering)<br>Anthropogenic (e.g. mining, burning fossil fuels, landfill, waste incineration and agriculture) | Soil guideline values (Clea, 2009)<br>Little evidence of thresholds being exceeded in Scotland. | Data from National Soil Inventory for Scotland (NSIS1978-88 & 2007-09).<br>Data from primarily agricultural soils held by SRUC.<br>UK Soil & Herbage pollutant Survey (UKSHS) (Total of 40 rural, 6 Urban and 10 industrial sites in Scotland, Environment Agency 2007) |
| Polychlorinated Biphenyls (PCBs) | Industrial and manufacturing processes | USEPA (PCBs: 10mg/kg), not exceeded | NSIS2007-09, 4 transects across Scotland (total of 30 sampling sites),<br>UK Soil & Herbage pollutant Survey (UKSHS) (Total of 40 rural, 6 Urban and 10 industrial sites in Scotland, Environment Agency 2007) |
| Polycyclic aromatic hydrocarbons (PAHs) | Burning of fossil fuels | USEPA (e.g., Benzo (a) pyrene: 2.9mg/kg), not exceeded. | NSIS2007-2009, 4 transects across Scotland (total of 30 sampling sites),<br>UK Soil & Herbage pollutant Survey (UKSHS) (Total of 40 rural, 6 Urban and 10 industrial sites in Scotland, Environment Agency 2007). |
| Pesticides | Primarily agricultural activities | USEPA (Atrazine: 110mg/kg), not exceeded. | UK pesticide monitoring programme in place since 1985 but seldom done in terrestrial environment.<br>Scottish data only for 1 catchment (Ugie). |
| Antibiotics Resistance Genes | Primarily livestock agriculture. | Unknown | Data from NSIS 2007-09 samples evenly distributed across Scotland (Pagaling *et. al.* 2023) |
| Microplastics (plastic particles less than 5um in size) | Almost every industry | unknown | NSIS2007-09 samples currently under analysis |

and leaching are typically small. This would also be likely for other chemicals, particularly for the organic pollutants (e.g., PCBs, PAHs) which have hydrophobic properties. Initially these compounds could potentially leach into the waters but over time, these compounds tend to become bound to the soil organic matter and the likelihood of leaching becomes reduced and they may be associated more with runoff. In Scotland only one study exists that investigates the transport of emerging contaminants between soils and waters (Cui *et. al.,* 2020a).

### 3.3.2 Costs of soil contamination

Soil contamination costs can be estimated from the cost of returning the contaminated soil to a usable state. Given the large variation in organisations or persons responsible for carrying out this remediation, as well as differences in types of contamination, the intended use for the land on which soil has been contaminated, and landscape and soil variables it is not possible to estimate the total costs of remediation of all of Scotland's soil contamination. Some estimate of the scale of costs could be estimated from the current remediation carried out for contaminated land. The costs of remediation also vary significantly, from £24 per cubic metre for in situ natural attenuation to £1,067 per cubic metre for offsite incineration (Summersgill, 2006). To provide any estimate of costs for soil contamination it would therefore be necessary to first understand how remediation would be carried out for the emerging contaminants, and impacts on the services provided by soil, including the potential for a loss of the soil resource if it needs to be removed.

Soil contamination also has potential wider societal costs. Contaminated soil may be unsuitable for crop growing, or impact crop yield (Shahid *et. al.,* 2015, Zhao *et. al.,* 2022), with direct economic impacts. Livestock may also be impacted, either directly through negative impacts of contaminants, or through bioaccumulation rendering the meat inedible (Rajaganapathy *et. al.,* 2011, Rhychen *et al.,* 2014). Contamination entering watercourses can have significant ecological impacts, as well as on drinking water quality, leading to increased costs of water cleaning. Finally, contaminants can have impacts on physical and mental health (Beseler *et al.,* 2008).

# 4 Discussion and Conclusions

This project has developed a framework and outlined the data available to:

- Estimate the effects of soil degradation (compaction, sealing and contamination) on soil properties.
- Calculate the extent of these degradation processes.
- Assess the wider impacts on soil functions.
- Cost associated impacts on the loss of soil functioning.

Soil compaction alone already costs farmers more than £25 million per year due to yield loss and additional fuel use. The cost of worsening compaction could exceed £70 million at the farm gate, without considering more inefficient fuel use in compacted soils. The overall costs of soil degradation to the wider Scottish economy are far greater due to impacts from erosion, flooding, contamination, biodiversity loss and greenhouse gas production.

The costs presented here are minimum estimates, based on a limited number of impacts for which both economic and biophysical data are available. We have only considered market costs, and have not considered preferences for protection, which will likely greatly increase the costs of impact, such as flood mitigation. Costs are mostly presented on a per unit basis due to biophysical data not being modelled in a way that fully quantifies impact associated costs (Table 9).

Additionally, the costs presented are not spatially differentiated, with a single per unit cost applied across Scotland. This is primarily driven by data on spatial differences in costs being highly complex. Analysing, and using, spatially differentiated costs combined with improved certainty in the assessment of the biophysical impacts would provide an understanding and allow the protection priorities for soil resources to be identified.

Costing data is not often specifically related to soils with additional modelling required to link soil degradation to the available associated costing data. Due to this, there is greater uncertainty in attributing costs to specific degradation processes.

## 4.1 Compaction

The extent of soil compaction was obtained through an analysis of available point data

**Table 9 Total costs of soil degradation in Scotland.**

| Degradation process | Element costed | Scottish Extent | Price per unit | Total cost |
|---|---|---|---|---|
| **Soil contamination** | Remediation – soil treatment | Unknown | £24 - £1,067 per cubic metre treated | Unknown |
| **Sealing** | Flooding | Number of properties impacted - unknown | £56,776 - £76,035/property in building damage and contents per household | Unknown |
| | | 0.04% increase in run-off per $km^2$ sealed. | £259 million annual damages reported by local authorities | £2.6 million for a 1% increase in flooding |
| | Remediation - SUDS | Number and types of SUDS installed - unknown | £69,000 - £273,000 per SUDS over 50 year lifespan. | Unknown |
| **Compaction** | Flooding | Number of properties impacted - unknown | £56,776 - £76,035/property in building damage and contents per household | Unknown |
| | | Increase in flood events or flood area due to compaction - unknown | £259 million annual damages reported by local authorities | £2.6 million for a 1% increase in flooding |
| | Crop losses | 109,479t - 324,217t barley lost | £150/tonne – barley farm price | £16.4m - £48.6m Scottish total |
| | Additional fertiliser[1] | Run-off in t – unknown | N - £2,430/t<br>P - £2,010/t<br>K - £1,280/t | Unknown[1] |
| | Additional fuel | 29% (sandy soils) to 87% (clay soils) increase. | Red diesel - £0.89/l | £15-£209/ha/yr<br>£9-26 million Scottish total |

[1] The £10.1million nutrient replacement cost from losses due to erosion of calculated in Rickson *et. al.* 2019 is likely to be increased by soil compaction and increases in fertiliser costs.

on topsoil air-filled porosity from 4 data sets that showed 56% of the land classified as at high vulnerability to topsoil compaction was compacted. Additionally, 21% of land classified as moderate and 12% at a low vulnerability to compaction, was compacted. This allowed some limited spatial representation of the likely extent of compaction in Scottish soils.

Methods were developed to further characterise compacted soil profiles under cultivation, and these have been applied to each representative soil series profile data for modelling both the impact of compaction on crop yield and on runoff (Figure 12).

Results from the DSSAT crop model, simulating crop growth for compacted and uncompacted soil, shows compaction has a direct impact on crop yields and associated potential losses of up to 18.2%. Using Spring barley as a representative crop type, the yield penalty was estimated at between £16 million (current extent of cultivated land compacted) and £49 million per year (assuming all cultivated land was compacted). This is a direct cost to agribusinesses with the greatest yield losses occurring in soils with the lowest water holding capacity and/or in climatically dry areas. The model has also shown the potential for additional yield penalties due to the changes in nitrogen dynamics within compacted soils.

An approach to modelling the impacts of compaction on runoff was further developed following on from previous work (Hallett *et. al.,* 2016). The modelled proportionate loss in soil porosity due to compaction was used to amend the soils Standard Percentage Runoff (HOST SPR), (used in national flood modelling). These amended values were then applied to seven catchments and show an increase in Catchment SPR of between 0.2% to a change of 2% (current extent of cultivated land compacted) and from 1.2% to 7.8% (All cultivated land compacted). Of the seven catchments studied the greatest percentage changes are seen in the Ugie and Pow catchments where there is the greatest proportion of agricultural land.

This is one of the first attempts at translating nationally available data sets and calculating runoff based on water retention characteristics but there remains a large amount of uncertainty in the estimates of a compacted soils profile and the additional runoff from compacted soils.

Soil compaction alone may affect water quality by changing nutrient cycling, but for our estimates we considered costs of nutrient exports to water courses following those proposed by Rickson et al. (2019). With this, compaction will lead to nutrient exports by increased runoff. This loss of nutrients will also have direct costs in agriculture from fertiliser use efficiency.

Beyond those costs applied to the available data in this study, other data and costs associated with degradation of water quality data for Scotland are lacking. All recent research on costs of water, including the costs associated with erosion, use data from England and Wales for the 1992–1997 period, as first presented in Pretty *et. al.* (2000). The true costs of the impact of soil degradation on water quality in Scotland are therefore unknown.

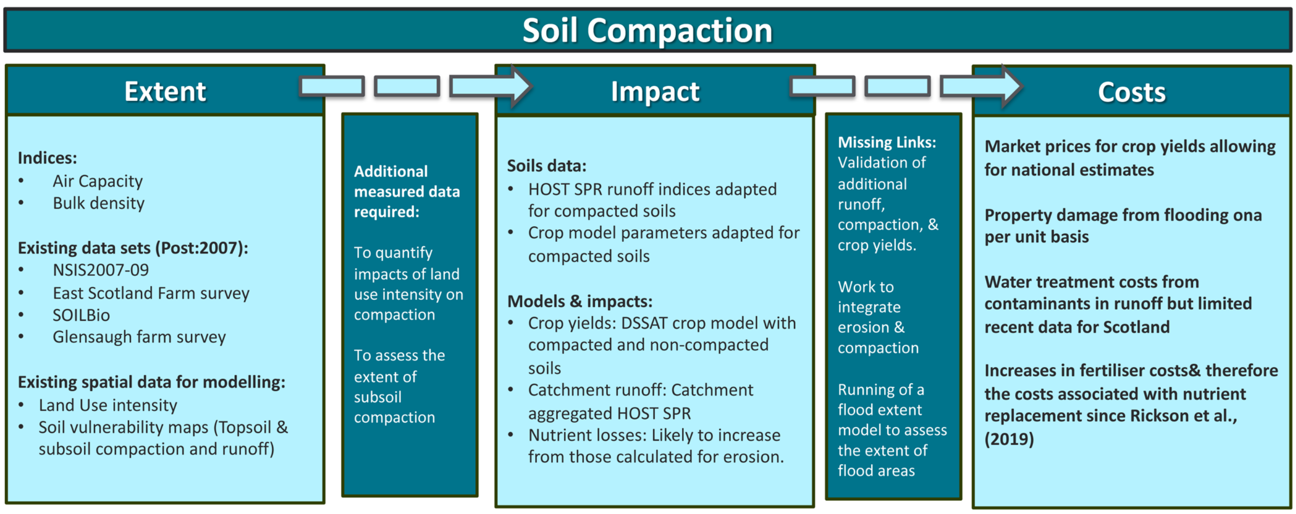


**Figure 12 Outline of the extent, impacts and costs of soil compaction and the additional data and modelling needed to link extent to cost impacts in Scotland.**

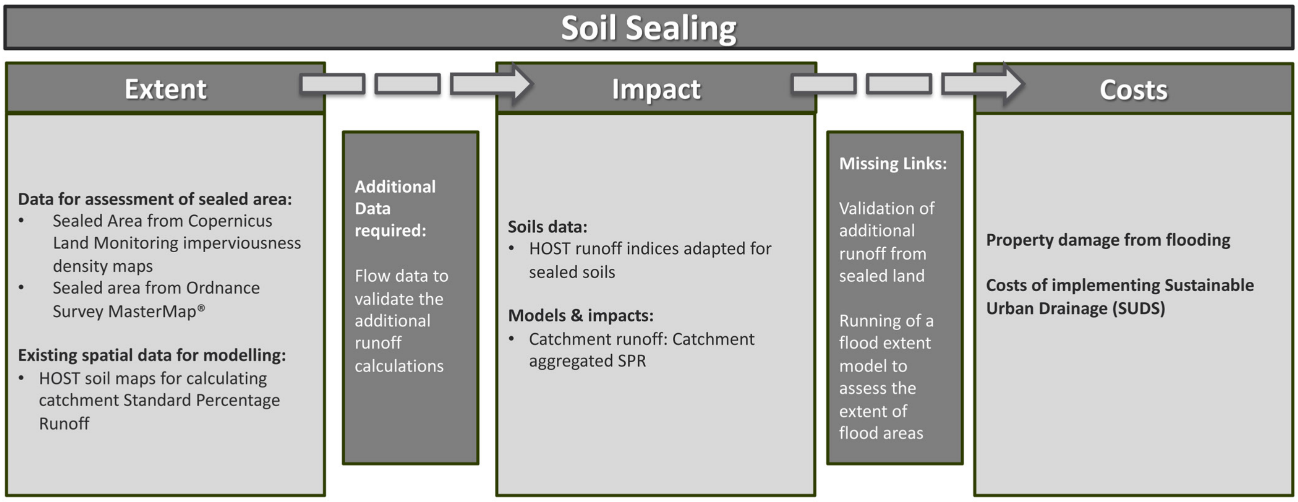


**Figure 13 Outline of the extent, impacts and costs of soil sealing and the additional data and modelling needed to link extent to cost impacts in Scotland.**

## 4.2 Sealing

This project compared two methods for calculating the extent of soil sealing (Figure 13) and demonstrated the value of UK Ordnance Survey data and validated the approach used by NatureScot to calculate the 'Soil Sealing Ecosystem Health Indicator'. A method was then further developed using the Hydrology of Soil Types (HOST) classification of soils and catchment Standard Percentage Runoff (SPR) (used in National flood modelling) to calculate the additional runoff caused by sealing.

The increase in Standard Percentage Runoff (SPR) from the Dee and the Don catchments is less than 0.9%. However, at this scale of catchment, the runoff calculations are greatly influenced by the large areas of uplands in the catchments relative to urban and sealed areas. The increase in runoff for a sub-catchment of the Dee which included recent development of the Aberdeen Western Peripheral Route and expansion of Westhill urban area was more than 1.5%. These runoff calculations can be explored further by a more in-depth analysis of the SPR from sealed areas and the impacts at times of peak flows. This will be key in assessing the impacts on drain flows, river flows, and also pluvial flooding, which was outside the scope of this project.

## 4.3 Contamination

There are less data on the extent of contaminants and the thresholds at which a soil becomes degraded which makes it difficult to determine the wider impacts such as the loss of land for agriculture. The results presented here are not able to link existing knowledge on the extent of contamination to enable the calculation of associated costs (Figure 14) and a recommendation would be for further work in this area.

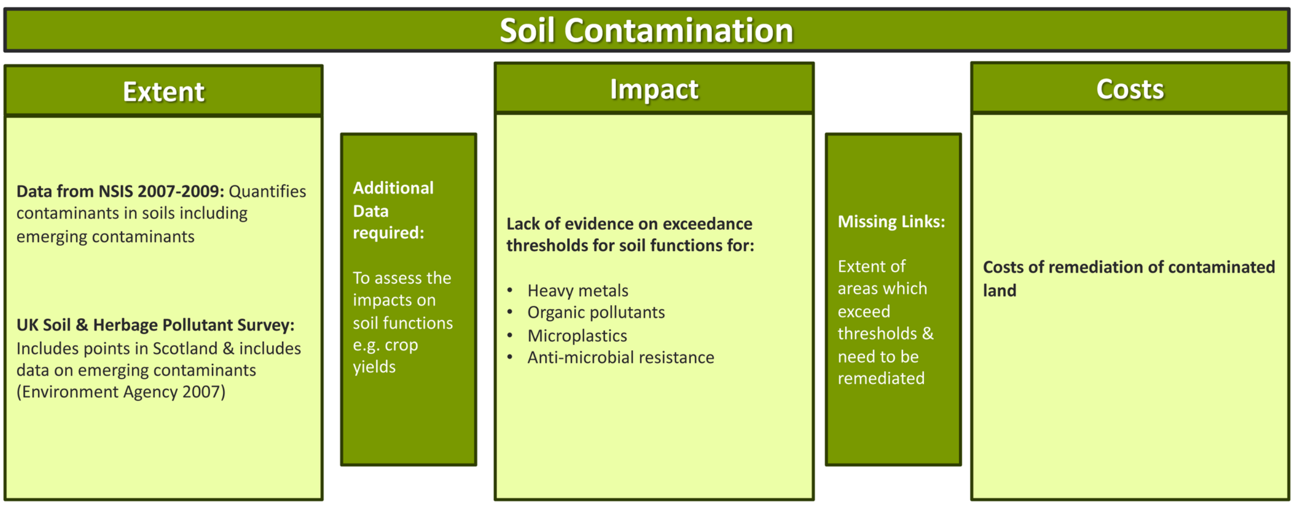


**Figure 14 Outline of the extent, impacts and costs of soil contamination and the additional data and modelling needed to link extent to cost impacts in Scotland.**

The potential impacts of soil contamination, for example on health and wellbeing, could be wide ranging making it challenging to determine which costs associated with them could be specifically linked back to soil contamination.

## 4.4 Future work

This project has demonstrated that all of the degradation processes considered in the report are a threat to Scotland's soils and have very large on and off-site costs. However, there is limited data available to quantify the extent or validate the modelling approaches needed to assess the impacts. There is also a requirement for integrated data on the extent, impact, and costs of degradation. Specific further work should include:

- A national field-based assessment of the extent of both topsoil and subsoil compaction to provide improved understanding of the relationships between erosion, runoff, and compaction. This requires a broader set of indicators including mechanical measurements to identify compaction and its impact on plant root growth.
- An assessment of topsoil and subsoil compaction in the built environment, with an aim to improve best practice in construction activities.
- Work to unpick the interrelationships and generate a new set of rules/models that allows the separation of the combined impact of compaction and erosion on nutrient exports to waters.
- An assessment of the structural condition of compacted soils that have been remediated to a given bulk density (e.g. through tillage operations) as this is likely to be poorer and the soils will therefore be more vulnerable to erosion.
- New data on GHG emissions to quantify the impact of compaction. Existing data for Scotland on arable and grassland soils are sparse and were inadequate for analysis of the impact of soil degradation on GHGs.
- Incorporation of the increased runoff from compacted and sealed soils into the Flood Estimation Handbook software to provide an estimate of the additional areas and infrastructure that may be impacted by an increased flood extent. This integration is possible as the additional runoff from sealing and compaction has been based on changes in HOST SPR which is the basis for the runoff calculations in the flood modelling software.
- Establishment of threshold values at which contaminants impact soil functions.
- Additional monitoring data on soil contaminants to establish the extent of contaminants in Scottish Soils, including data for wider emerging contaminants such as Per and polyfluoroalkyl substances (PFAS) for which there is currently no Scottish data available.
- Soil biodiversity is widely recognised as being crucial to the functioning of the soil ecosystem but also impacted by all the degradation processes reviewed. No extensive data sets on the impacts of compaction, sealing or contamination on soil biodiversity exist for Scotland. A list of practical, useful metrics of soil biodiversity which could be monitored across Scotland is required in addition to establishing whether past publicly funded small-scale datasets containing these metrics can be merged to establish a baseline estimate.
- Development or refinement of conceptual model of the link between soils and socio-economic systems, enabling wider impacts to be explored.
- Coordination of soil science and socio-economic data collection, measuring impacts across the same spatial and temporal scale to improve integration, and enable national level spatially explicit costs to be estimated.

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# 6 Appendix 1 – Reviews

## 6.1 Review of soil compaction in Scotland

Soil compaction has on-site economic costs due to decreased land productivity and poorer resource use efficiency, and off-site economic costs due to environmental pollution (groundwater, surface water and greenhouse gases) and soil erosion (Chamen *et al.,* 2015) (Figure 15). Many different land use systems can be affected by soil compaction (Ferreira *et al.,* 2022). Farming, which covers about 80% of Scotland's land, can cause soil compaction by the weight of machinery or livestock (Romero-Ruiz *et al.,* 2023). Likewise, compaction can result from heavy machinery used in forestry combined with sensitive soils (Nawaz *et al.,* 2013, Nazari *et al.,* 2021). Amenity surfaces and recreational areas are affected by people trampling (Schmid *et al.,* 2017). Construction, mining and installing utility corridors also cause compaction (Worlanyo and Jiangfeng, 2021, Brehm and Culman, 2023). While major infrastructure projects aim to minimise and mitigate compaction damage (Thompson *et. al.,* 2022), for house building the extent of this threat and its implications have been poorly explored.

The amount of land in Scotland affected by soil compaction is unknown and attempts to model soil compaction vulnerability, exposure and risk have identified considerable uncertainty (Troldborg *et al.,* 2013). While quantitative data exist on soil erosion and organic matter across the globe, much less exists for soil compaction (Kibblewhite *et al.,* 2016). Maps of soil compaction vulnerability for Scotland indicate that much of its land is vulnerable, especially to subsoil compaction (https://map.environment.gov.scot/Soil_maps/?layer=17#). The extent and effects of soil compaction have been exacerbated by more erratic weather that has been observed in recent years. Likewise, trends of increased machinery weight have increased stresses on soil (Keller *et al.,* 2019), causing compaction to deeper layers in the subsoil that is difficult to mitigate (Keller and Or, 2022) (Figure 16). A compacted soil holds less water and restricts root growth, so productivity under dry conditions can decrease markedly (McGeary *et al.,* 2022) or require more irrigation. The soil is more difficult to cultivate, so more fuel is used for tillage, and the days available for field operations can be limited if compacted soils drain more poorly.

Land managers experience a direct economic cost from compaction when undertaking mitigation practices (Chamen *et. al.,* 2015). Options, such as subsoiling, are an extra cost of labour, fuel and machinery. The effectiveness of subsoiling is variable and can be short-lived, possibly leading to greater subsoil compaction of a loosened matrix of previously stable soil (Chamen *et. al.,* 2015). A previous economic study on the impacts of soil compaction found that compaction avoidance was

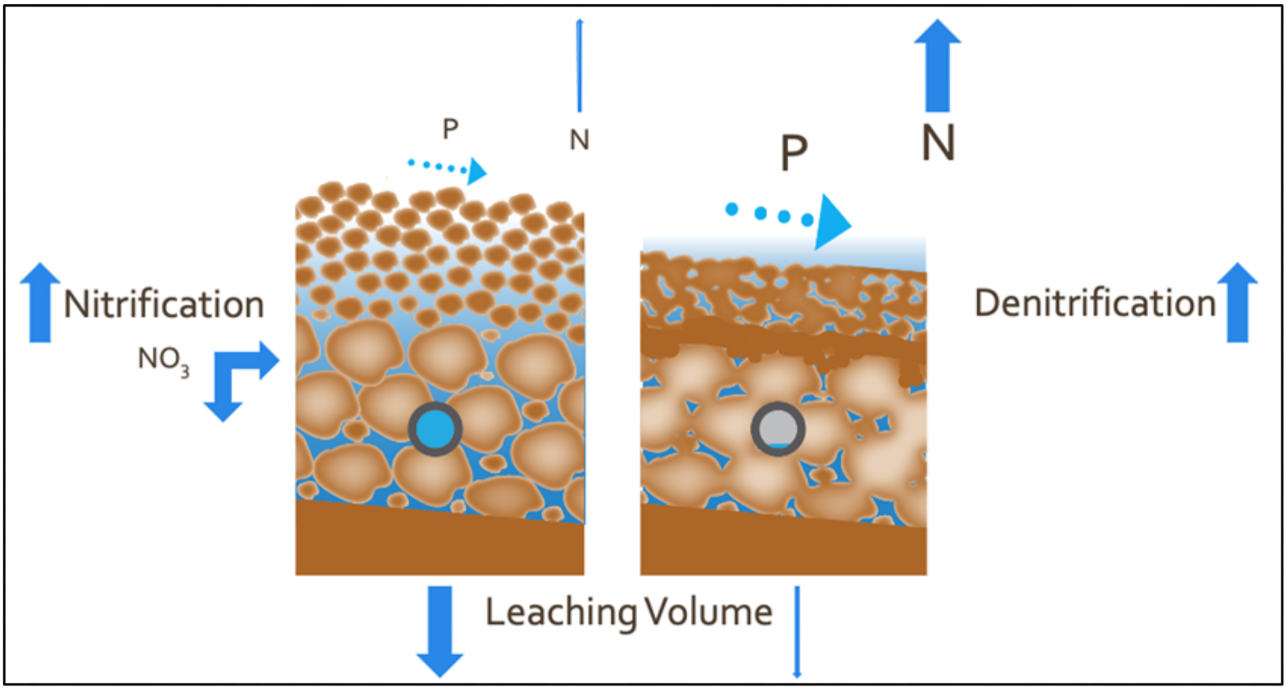


**Figure 15 - Nutrient losses increase due to compaction.**

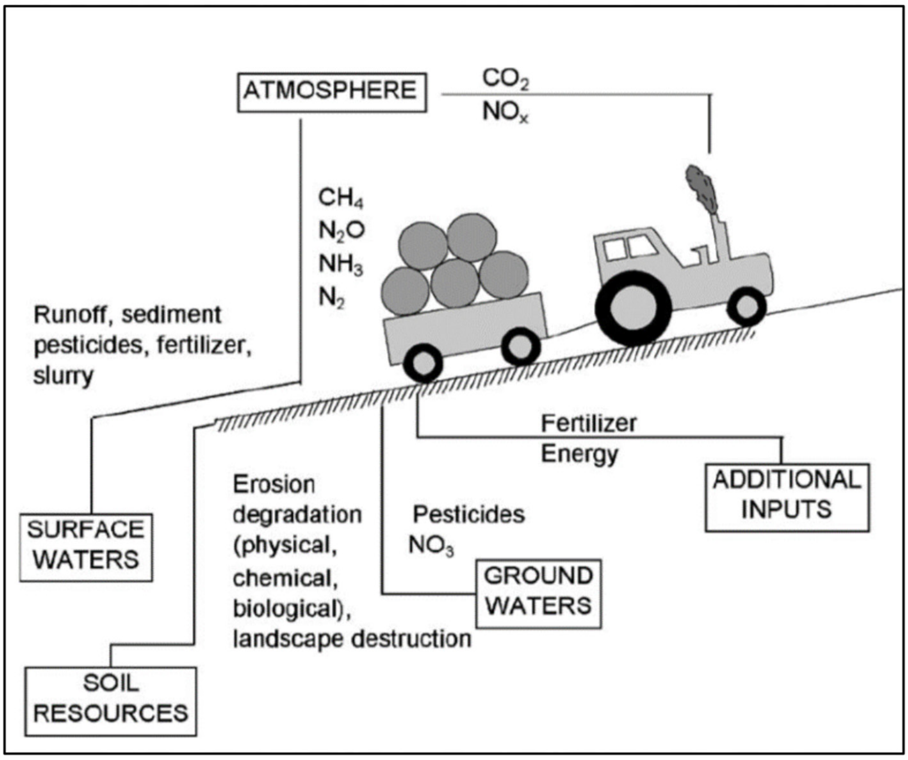


**Figure 16 Impacts of soil compaction on a range of ecosystem services in including ground and surface waters, soils and the atmosphere. This diagram omits the potential negative implications to crop productivity. (Soane and Van Ouwerkerk, 1995).**

much more effective than mitigation for direct costs at the farmgate. The same would hold for forestry or construction activities. By far the best practice to avoid soil compaction is to avoid traffic when the soil moisture is too high (Jones *et. al.,* 2003). Changed precipitation patterns and an increasing reliance on farm contractors, however, have constrained the capacity to time activities effectively. Low ground pressure tyres and tracks exert less pressure on the topsoil, so the window of opportunity increases, but this still presents a risk of subsoil compaction.

Whereas increasing machinery weight trends have adverse impacts on soil compaction, a move towards conservation tillage and new technologies can have positive impacts. Limiting the area of trafficked land is one of the best soil compaction avoidance approaches. Whereas agricultural machinery used to traffic up to 95% of ploughed land in a given year, one study found it decreased to about 70% under reduced tillage and 55% under zero tillage (Kroulik *et. al.,* 2009). This is promising as soil compaction is a serious concern with conservation tillage practice. However, one study found a 15% reduction in grain yield due to soil compaction of a zero tillage versus ploughed soil (Salem et al., 2015). A shift towards reduced tillage can lead to increased bioturbation that ameliorates soil compaction, but this requires the right site conditions that are favourable to earthworms and deep rooting crops in the rotation (Schlüter *et. al.,* 2018). UK studies exploring reduced tillage (shallow non-inversion to 10 cm depth) found that the economic cost of a minimal yield penalty was offset by savings on fuel and labour (McKenzie *et. al.,* 2017). In Sweden, under similar climatic and soil conditions to Scotland, compaction under reduced tillage was found to have only a minor effect on cereals, but dicots were more sensitive (Arvidsson *et. al.,* 2014). This helps explain the 10% adoption of conservation tillage practices by Scottish farmers (Scottish Survey of Farm Structure and Methods, 2016), but these levels are far behind global adoption. Between 2013 and 2016, the percentage of land under zero and conservation tillage almost halved.

Regardless of tillage practice, the advent of satellite guided machinery has improved the ability to constrain traffic to defined tramlines. However, different axle widths between machinery used for different farming activities can still result in compaction to less than 20% of a field. Controlled-Traffic-Farming (CTF) systems, where axle-widths are unified between different machinery types, have been advocated to avoid soil compaction. One form of CTF is a gantry system that supports implements used from soil tillage to harvest, but uptake of these systems by farmers has been slow in Scotland due to farm structural limitations and the cost of new machinery. In forestry, defined traffic routes are used as best practice, sometimes supplemented with brush mats to spread out stress transmission. Lighter vehicles operating as swarms have been advocated as another soil compaction avoidance approach. For harvesting equipment, it was estimated that 6-9 small (50 kW) harvesters accessing unloading facilities about every 3 minutes would be required to replace one conventional 10-20 Mg (300W) harvester (McPhee *et. al.,* 2020).

Another mitigation strategy is the planting of leys, which relax the frequency of machinery passes and can break up soil by the action of plant roots (Muhandiram *et. al.,* 2020). A ley-arable rotation has benefits beyond soil compaction, including increased soil carbon, biological activity and nutrient available from N-fixing or P-releasing plants. Economic losses from the land being out of crop production are offset by the value of the ley land for grazing or silage, and improved soil and nutrient conditions that many benefit follow-on crops.

Farmers are generally aware of some of the on-site economic costs of compaction. An online survey in Germany found that 85% of farmers adopt agronomic practices to avoid compaction, such as 78% who have altered tyre or chassis width (Ledermuller *et al.,* 2021). More heavily trafficked regions of farms, whether as access/ turning locations for machinery or intense grazing locations like feed troughs, have visually apparent stunted and less abundant plants that are more prone to moisture stresses. Mitigation options are therefore commonly practiced. Topsoil compaction mitigation is achievable by tillage, but the soil structure may still remain degraded. Subsoil compaction can be mechanically loosened by the deep tynes of a subsoiler, which is common practice in Scotland. However, the impacts can be short-lived and the disturbed soil may be even more susceptible to subsequent compaction (Chamen *et al.,* 2015).

Greater fuel use from subsoiling or cultivating compacted topsoils produce greenhouse gas emissions that have economic costs to the wider environment (Zabrodskyi *et. al.,* 2021). Poorer fertiliser use efficiency results in compacted soils due to constrained root growth, anaerobic conditions, and increased overland flow. Anaerobic conditions transform nitrogen into less accessible and more mobile forms, producing toxic nitrite that contaminates groundwater and greater greenhouse emissions of nitrous oxide (Pulido-Moncada *et al.,* 2022). To compensate, farmers may inadvertently apply more fertiliser. Phosphorus bonds strongly to soil particles, so it is lost from the increased erosion of less permeable compacted soils.

Consequently, in addition to the on-site economic costs of poorer fertiliser use efficiency, off-site environment costs result from greenhouse gas emissions and water contamination. From a meta-analysis, emissions of nitrous oxides in compacted soils have been found to nearly double in arable and pasture soils, and increase five-fold in forest soils (Hernandez-Ramirez *et al.,* 2021). The costs of these diffuse forms of pollution are very difficult to estimate. There are other off-site costs from soil compaction that have off-site environmental costs, but an economic assessment would be extremely speculative with existing knowledge (Chamen *et. al.,* 2015, Table 9 and 10). This includes impacts to biodiversity, such as microbial habitat space in soils or on feeding birds that peck the surface in search of soil organisms. A vicious circle arises if suppressed biological activity due to soil compaction, suppresses biological activity like earthworm burrowing or root growth that can ameliorate soil compaction (Meurer *et. al.,* 2020). There is anecdotal evidence of a link between flood risk and soil compaction. In urban environments, decreased infiltration of water into compacted soils has been found to increase runoff and flooding considerably (Yang and Zhang, 2011).

| Table 10 Update of Chamen *et. al.* (2015) providing an overview of literature comparing crop yield on trafficked versus non-trafficked plots. A range of soil types, crops and countries are presented, with data most relevant to Scotland highlighted in bold (See Chamen *et. al.* (2015) for references). | | | | |
|---|---|---|---|---|
| **Crop** | **Yield (% of non-trafficked)** | **Soil Information** | **Country** | **Reference** |
| Cereals | 87-110 | Profile: clay, loam, sandy loam, loam | England, Netherlands, Scotland, Germany | Chamen *et al.,* 1992b |
| Barley | 62-81 | Subsoil: sandy loam | England | Pollard & Elliott, 1978 |
| Wheat | 85 | Profile: clay | England | Chamen *et al.,* 1992a |
| Spring barley | 86 | Profile: clay | England | Chamen *et al.,* 1994 |
| Wheat | 79 | Profile: clay | England | Chamen & Longstaff, 1995 |
| Wheat | 100 | Profile: silt loam | England | Graham *et al.,* 1986 |
| Barley | 100+ | Profile: sandy clay loam | Scotland | Campbell *et al.,* 1986 |
| Spring barley | 84 | Profile: gley[2] | Scotland | Dickson & Ritchie, 1996 |
| Spring osr | 80 | Profile: gley[2] | | |
| Winter barley | 87 | Profile: gley[2] | | |
| Wheat | 74 | Raised beds: sands, loams | Australia | Hamilton *et al.,* 2003 |
| Barley | 69 | | | |
| Oilseed rape | 75 | | | |
| Wheat | 83 | Profile: clay loams | USA | Voorhees *et al.,* 1985 |
| Maize | 79 | | | |
| Soybean | 84 | | | |
| Wheat | 93 | Profile: loam | Netherlands | Lamers *et al.,* 1986 |
| Cereals | 69 | Profile: clay loam | Australia | Radford & Yule, 2003 |
| | 80 | Subsoil: clay | | |
| Cereals & grain legumes | 89 | Profile: Red Brown earth | Australia | Sedaghatpour *et al.,* 1995 |
| Wheat | 100 | Profile: clay | Australia | Radford *et al.,* 2000 |
| Wheat | 84 | Profile: fine sand | South Africa | Bennie & Botha, 1986 |
| Cereals | 87-95 | Profile: various | Ukraine | Medvedev *et al.,* 2002 |
| Cereals | 77–122 | Profile: various | Poland | Lipiec, 2002 |
| Oats | 71 | Profile: clay | Sweden | McAfee *et al.,* 1989 |
| Barley & peas | 77-100 | Subsoil: silt loam | USA | Hammel, 1994 |
| Wheat | 100 | | | |
| Oilseed rape | 53 | Profile: sodic clay | Australia | Chan *et al.,* 2006 |
| Spring cereals | 79-84 | Profile: clays | Sweden | Håkansson *et al.,* 1985 |
| Cereals | 88 | Profile: clay | Australia | Tullberg *et al.,* 2001 |

| Table 11 Yield reductions observed in field studies comparing compacted to less compacted soils. | | | | |
|---|---|---|---|---|
| **Crop** | **Yield (% of non-compacted)** | **Soil Information** | **Country** | **Reference** |
| Cereals | 95 | Profile: < 17% clay and >1.82 g gm-3 PD | Germany | Schneider and Don, 2019 |

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## 6.2 Review of soil contamination in Scotland

### 6.2.1 Introduction

Soil is an important matrix which is a mixture of organic matter, minerals, gases, liquids, and organisms that together support the life of plants and soil organisms. Therefore, it exerts critical role to promote sustainable ecosystem in aspect of resource conservation, energy storage, wide biodiversity, climate change, and human wellbeing. However, a wide range of soil contamination is increasing global concern due to expansion of anthropogenic activities (i.e., agricultural and industrial production etc.) which intensify environmental degradation.

Soils are rich in organic carbon or organic matter, which provides favourable conditions for the sequestration of chemicals and has been shown to be a sink for many different chemicals. Soil ecosystems are experiencing unexpected disturbances as a result of increased human activity and the pressures of climate change. As a result, it is important to understand the contaminants in soils and their potential threat to the soils.

Scottish government issued the Scottish Soil Framework to promote sustainable management and soil conservation in response to economic, social and environmental needs of Scottland. The Scottish Government published the Scottish Soil Framework which listed the top 10 threats to Scottish soils and noted that soil contamination by heavy metals can be locally significant and other contaminants including persistent organic pollutants and pesticides need to be considered in future (Scottish Government, 2009; SEPA, 2014).

In this report we review the levels of heavy metals, classical and emerging organic contaminants including PAHs, pesticides, antibiotics, anti-microbial Resistance, and microplastics (MPs) in soil matrices across Scotland. Additionally, historical trends in the evolution of pollutants in Scotland are summarized with a view to better understanding the changing characteristics of these pollutants in response to human activities and policy interventions.

### 6.2.2 Contaminants in Scottish Soils

**Heavy metals**

Heavy metals are a group of elements that are widespread in the natural environment. Trace amounts of metals provide favourable conditions for the growth and reproduction of soil organisms and plants, but once thresholds are exceeded, their growth is adversely affected, and they might be enriched through the food chain, ultimately threatening human health. Potential sources of contamination from locally high concentrations of heavy metals included both natural sources (e.g., volcanic eruptions, weathering of rocks) and anthropogenic sources (e.g., mineral exploitation, fossil energy combustion, landfill or incineration and agricultural practices). From the mid- to the late 1970s, the soil in the region of Armadale, Scotland, was contaminated with heavy metals because the foundries were the main industry in the area, with arsenic concentrations in the ranges of 7–63.9 ppm (mean: 18.2 ppm) higher than those in Whitburn (4.6-11.6 ppm, mean: 8.1 ppm) where the coal industries were the main industry (Smith et al. 1986). Elevated levels of Pb were found in cores from highland catchments in North East Scotland, with concentrations ranging from 52±13 to 258±13 mg/kg (Farmer *et. al.* 2005). Concentrations of Cr, Cu, Ni, Pb and Zn in soils from central Scotland were distributed in the ranges of 39-102, 5-35, 10-52, 19-202, 19-161 mg/kg, respectively (Bacon and Hewitt 2005). Also, nationwide soil monitoring in Scotland in 2007-2009 showed Pb in mineral soils at concentrations ranging from 1.3 to 133 mg/kg versus 4.3 to 580 mg/kg in organic soils (Farmer *et. al.* 2016). Spatially, Pb levels in organic soils of central and southern Scotland had significantly higher concentrations than those in the north, while there was no significant difference in mineral soils. This is mainly attributed to the fact that Pb in organic soils is predominantly anthropogenic atmospheric deposition, whilst for mineral soils it is a combination of anthropogenic and natural Pb. These studies suggested that Pb might have relatively high levels of contamination in this area, with significantly higher contents than those in Scottish soils in 1984 (2.5-85 mg/kg) (Reaves and Berrow 1984), indicating that anthropogenic Pb has deposited more widely in the soil.

**Persistent organic pollutants (POPs)**

Persistent organic pollutants (POPs) are a class of semi-volatile, resistant to degrade, long-range transport, bioaccumulated and toxic synthetic chemicals (e.g., PCBs and PBDEs etc.). PCB is the most representative POPs because of its wide range of uses in industrial manufacturing, including coolants, capacitors, transformers, fireproofing materials, paints, etc. Historically, PCBs have been produced in the UK since 1954, and new uses of PCBs were banned in the UK in 1981. While PBDEs are brominated flame retardants and the use of PentaBDE and OctaBDE was banned in 2004 in the EU, USA and Canada due to concerns for human and environmental health (Shatalov et al. 2004). And the use of decaBDE in electronic and electrical equipment in the EU has also been further restricted since 2008.

Bracewell *et. al.* (1993) reported that PCBs in Scottish soil were significantly higher than previous soil studies in England and Wales, showing an increase in the total PCB concentration from north to south. Cachada et al. (2009) investigated the distribution of PCBs in the soils of five European cities, including Glasgow, where the topsoil (range: 4.5–78 µg/kg; median: 22 µg/kg for Σ19PCBs; range: 1.9–43 µg/kg median: 9.4 µg/kg for Σ5PCBs) was higher than the concentrations in other European cities (Aveiro, Uppsala, Torino and Ljubljan). This is mainly influenced by local sources of pollution and global atmospheric transport. Rhind *et. al.* (2013) determined the distribution of PCBs and PBDEs in Scottish topsoil for the years 2007-2009, with concentration distributions ranging from 0.04–11.2µg/kg for PCBs and 0.09-15.38 µg/kg for PBDEs. Zhang *et. al.* (2014) further compared the concentrations of PCBs and PBDEs in Scottish soils over a 20-year period (1990–2009), and showed a gradual decrease in PCB concentrations (4.91–57.7 ng/g in 1990, 0.23-21.4 ng/g in 1999, and 0.77–19.5 ng/g in 2007-2009) and a significant increase in PBDEs (0.02-1.57 ng/g in 1990, 0.41–10.5 ng/g in 1999, and 0.2–13.2 ng/g in 2007–2009) during this period, which also coincided with local policy/regulatory changes. PBDE concentrations in soil were significantly higher in the southern region than in the northern region, whereas there were no significant spatial differences in PCBs, and no significant further inputs of PCBs into the environment due to the ban on production, degradation, transfer and redistribution over time, which not only resulted in lower soil concentrations, but also eliminated the spatial differences in the level of contamination to a large extent. Compared to other regions, PCBs in Scottish soils were slightly higher than those in rural soils in Shanghai, China (mean: 515 pg/g) and lower than residential soils in East Chicago, Indiana (20–1700 ng/g). In terms of PBDE, the contamination levels in Scottish soils were close to those in the central Italian Alps (∑13PBDEs, 0.71±0.83 ng/g) and European background soils (0.065–12 ng/g), but significantly lower than those in soils near e-waste recycling sites, such as

Taizhou, China (824.4–948.6 ng/g). Overall, PCBs and PBDEs were at low levels of contamination in Scottish soils.

### Polycyclic aromatic hydrocarbons (PAHs)

Polycyclic aromatic hydrocarbons (PAHs) are also a group of toxic organic pollutants with long-range transport properties that are widespread in the environment, which can be from natural and anthropogenic sources. 16 PAHs have been identified as priority pollutants by the European Union (EU) due to their bioaccumulation and threat to human health. The contamination profile of PAHs in Scottish soils was mapped over the period 1990 to 2009, and the average concentrations of the 16 PAHs in 1990, 1999, and 2009 showed a generally decreasing trend, with 3659 ng/g, 1644 ng/g, and 727 ng/g, respectively (Cui *et. al.,* 2020b). And the mean concentration in the NSIS 2007–2009 soils was 1466 ng/g (Rhind *et. al.,* 2013). The ∑PAH16 concentrations in the three separate years in Scottish soils were comparable to contemporary arable, grassland and background soils from Poland, Norway, and China. However, concentrations of PAHs in this study were much lower than urban soils in Ljubljana (Slovenia) and Torino (Italy), respectively. Meanwhile, the range of soil PAH concentrations in Scotland is similar to those measured along an 80 km urban–rural transect in the Greater Toronto Area (Canada). The changes in concentrations of PAHs in Scottish soils at spatial-temporal scale may be driven by a comprehensive influence of emission source, population, climate change (temperature), latitude, SOC and land use type. Generally, the concentrations of PAHs in soils related to land use type and their levels in arable soils tend to be lower compared to other land uses such as grassland or forest soils. It is suggested that agricultural production and management practices result dilution effects for many pollutants (Cui *et. al.* 2020b).

### Pesticides

Pesticides play an important role in controlling pests and diseases, protecting crops and increasing food production, while their adverse effect on biodiversity, ecosystems and public health have also caused concern. The Pesticide Monitoring Program (PMP) has been introduced in the United Kingdom (UK) since 1985, which has supported the monitoring of a number of watersheds in the UK. However, there are not many works in terrestrial environment.

So far, only one paper on pesticides of Scottish soil reported (Cui *et. al.,* 2020a) the distribution of pesticides in the Uige River Catchment with the concentration levels in soil ranged from 1.7 to 8.0 ng/g (with a mean value of 4.7 ng/g), reflecting soil runoff as a potential source of pesticides to the river's water environment. It seems low level of pesticides in soils of Ugie catchment, however, it is difficult to drive conclusions on the level and impact of pesticides contamination in Scottish soils due to lack of dataset. It is suggested to have more soil monitoring work on such contaminants which would help to understand and assess their environmental impact.

### Antibiotics and Antimicrobial Resistance (AMR)

Antibiotics, the class of chemicals most closely associated with human life and agricultural activities, have a prominent role in the prevention and treatment of microbial infections, which has contributed significantly to the development of industries such as pharmaceuticals, modern agriculture and animal husbandry. The widespread use of antibiotics may exacerbate bacterial resistance and promote the spread of resistance genes, reducing the therapeutic potential of antibiotics for human and animal diseases. Antibiotic resistance is a global public health crisis. In 2019, the UK published the 20 Year Vision for AMR and a new 5-year national action plan, Tackling AMR 2019–2024, with the aim of effectively controlling AMR by 2040.

A paper has recently been submitted, by a James Hutton Institute Researcher, on AMR in the NSIS2007-09 samples. The work suggests that the risk for AMR in Scottish Soils appears to be low (Pagaling *et al.,* 2023). However, this may be due to the high proportion of sites with semi-natural land cover in the NSIS2007-09 soils. There is therefore a need to establish additional national monitoring data on these contaminants in intensively cultivated or contaminated soils to further understand their distribution in Scotland. Soils where sludge or manure are applied could be particularly at risk of contamination by AMR and could provide a focus for monitoring.

### Abstract for paper on AMR in NSIS2007–09 soils

*"Antimicrobial resistance (AMR) has been increasing through the use and misuse of antimicrobials, including antibiotics, threatening our drug therapies' effectiveness for infection control. The environment plays a significant role in*

*disseminating AMR genes, exacerbated by anthropogenic activities such as industry and farming. However, it is difficult to distinguish between what AMR is promoted or amplified by human activities and what is natural. Extending the scale and depth of monitoring efforts will allow a better understanding of the drivers of AMR, including what "background" resembles. Here, we quantified approximately 300 AMR-related genes in over 200 soil samples evenly distributed across Scotland (i.e., without considering possible AMR sources). Increases in ambient soil nutrient levels (e.g., natural organic matter and nitrogen) significantly reduce AMR gene richness (p < 0.01 for five gene classes), possibly due to a proliferation of susceptible bacteria out-competing resistant bacteria. However, locally elevated levels of some heavy metals (e.g., aluminium, barium and manganese) influence AMR gene richness but not AMR gene abundance. Persistent organic pollutants also increase transposase relative abundance, possibly promoting conditions conducive to the horizontal transfer of AMR genes. Although natural conditions impact local AMR prevalence, local AMR gene abundance and richness vary spatially according to resistance class, and humans can increase AMR in the environment, especially gene richness."*

**Microplastics**

Microplastics (MPs) are a class of plastic particles less than 5 µm in size that are widespread in freshwater and terrestrial environments around the world because of their small size and ease of dispersion. These substances are difficult to degrade, which allows them to persist in the environment for hundreds or even thousands of years and can be enriched through the food chain. The importance of understanding the fate and impact of these plastics in the environment has not been recognized until recent years.

A PhD researcher from James Hutton Institute is working on NSIS2007-09 samples for microplastics, however, the data is under analysis and the results can be shared once they have been published.

Studies on MPs in England have found them to be present in soils. For example, investigations of MPs in central and southern England found that MPs were detected in landfill soils (mean: 12.3 ± 27.5 MP/g), urban roadside soils (mean:17.3 ± 24.1 MP/g) and urban parkland soils (mean:15.7 ± 19.5 MP/g), whereas they were not detected in woodland soils (Billings *et. al.,* 2023). The intensity of anthropogenic activities may have contributed to these differences. However, mean MPs concentrations in agricultural soils from the Test catchment in the UK were high in both biosolids-treated sites (874 MP/kg) and untreated sites (664 MP/kg), with no significant differences between treated and untreated soils (Radford 2023). The sources of MPs contamination are diverse and might be stressed by numerous environmental factors.

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## 6.3 Review of degradation processes and carbon losses in peat and peaty soils

Peat and peaty soils store cover 64% of Scotland and store 72% of its topsoil carbon (2049 Mt in upper 1m (Rees *et. al.,* 2018)). With the notable exception of lowland raised bogs, peat soils are primarily located in the uplands of Scotland and primarily associated primarily with blanket bogs habitats while peaty soils (where the surface peaty layer is less than 50cm thick) are more likely to be found under heather or grass- dominated moorlands. These upland moorlands are used for a mixture of grouse and deer hunting and upland extensive livestock grazing and have suffered heavy degradation of the habitats and soils partially as a result (although long term natural processes play a part in some degradation). Below we will outline the primary soil degradation processes and losses of carbon from peat and peaty sils in Scotland.

### 6.3.1 Overview of Peat degradation

Peatlands is the generic term that describes wetland ecosystems associated to soils which are saturated and hold large deposits of organic plant material in various degree of humification (peat). Chapman *et. al.* (2009) estimated that all peat soils in Scotland (including those under woodland, grassland or arable cultivation) store around 1620 Mt (56% of all Scotland's soil carbonstocks). Historically, peat has accumulated as a result of high-water table, and/or low temperatures and continuous vegetation cover of mosses, sedges, and shrubs, making peatand a long-term sink for carbon. However, around 75% of peatlands, that is areas of peat soils with semi-natural, often bog, vegetation are modified or degraded, causing potential high losses of GHGs (Greenhouse gases) (Evans *et. al.,* 2017).

Peatland degradation causes significant losses of carbon to the atmosphere, either directly as gaseous losses as microbial communities metabolise peat to $CO_2$, or indirectly as particulate organic carbon (POC) or dissolved organic carbon (DOC) lost into waterways and subsequently metabolised by the aquatic microbial community (Evans *et. al.,* 2013). The primary cause of gaseous losses of carbon from peat soil is drainage or any process that lowers the annual average water table below between 10–20 cm and sections of the peat microbial community can aerobically decompose carbon (Evans *et. al.,* 2021). Peatlands export POC and DOC to the aquatic system through their drainage networks, degraded peatland systems often directly or indirectly increase the extent and significance of internal drainage systems and accelerate the rate of loss into these networks (Evans *et. al.,* 2016).

The primary causes of peatland degradation are climate change and human activities such as forestry, overgrazing, burning, pollution, drainage and cultivation (Evans *et. al.,* 2017), with these factors often acting in concert (e.g. drainage of peat bogs was often required for forestry planting). Degradation switches peatlands from slight carbon sinks or neutral GHG balance to strong sources of carbon to the atmosphere. In total, GHG emissions from Scottish peatlands were estimated in 2013 to be 9-10 Mt $CO_2$e, per year (Evans *et. al.,* 2017).

### 6.3.2 Drainage of peatlands

In total, across Scotland, there are approximately 762,000 ha (39%) of peatland that is likely to have undergone drainage, either confirmed as having a history of drainage or with land cover that demands drainage as part of the practice (such as forestry or agriculture) (Evans *et. al.,* 2017). This makes drainage the primary form of degradation that Scottish peats suffer from on an area basis.

Any process that lowers the water table of a peatland will likely increase $CO_2$ emissions; across the UK and Europe, regardless of land use on peat soils, the overriding control on $CO_2$ emissions is the water table (Evans *et. al.,* 2021). Where average annual water table depth is below approximately 10 cm, net ecosystem flux of $CO_2$ in peatlands switches from being a sink (net uptake) to a source (net release) to the atmosphere (Evans *et. al.,* 2021). When taking $CH_4$ into account, every 10cm drop in the water table increases GHG loss to the atmosphere by 3 tonnes of $CO_2$e (equivalent GHG warming potential) per hectare per year (Evans *et. al.,* 2021).

Peatland drainage causes up to 100% more DOC losses (average of 60 % across all measured sites) compared to undrained peatlands in boreal, temperate and tropical bogs and fens (Evans *et.al.,* 2016). Conversion of DOC to $CO_2$ in stream is known to be rapid, especially in headwater streams where $CO_2$ evasion rates can be high (Dinsmore *et. al.,* 2010). At present, it is estimated that 90% of DOC that enters stream and river systems is lost to the atmosphere as $CO_2$ and $CO_2$ -equivalent ¬emissions from DOC (including methane) are around 1.05 $CO_2$e $ha^{-1}$ $yr^{-1}$ (Evans *et. al.,* 2016).

### 6.3.3 Erosion of peatlands

Eroding peatlands are a major source of $CO_2$ to the atmosphere, not only because erosion features/gullies cause drainage of peat (Artz *et. al.,* 2022), but because they have high loss of particulate organic carbon (POC) (Evans *et. al.,* 2013). Much of the erosion of peat soils found in UK peatlands has resulted from anthropogenic pressures, including burning (Yallop *et. al.,* 2009), overgrazing by deer and sheep (Worrall and Evans 2009, Werritty *et. al.,* 2015), artificial drainage installation (Holden *et. al.,* 2007, Worrall and Evans 2009) and atmospheric pollution (Yeloff *et. al.,* 2006).

Eroding blanket bogs cover around 273,000 ha of Scotland (Evans *et. al.,* 2017) and are potentially a strong direct source of $CO_2$ to the atmosphere at a rate of 3.6 t $CO_2$ $ha^{-1}$ $yr^{-1}$ (Artz *et. al.,* 2022). Due to the remote nature of blanket bogs, there are limited measurements of GHG exchange, however, the measurements by Artz *et. al.* (2022) are likely to be reasonably reflective of other eroding blanket bogs because their area of study had similar levels of erosion as the average eroding blanket bog (Evans *et. al.,* 2017). However, this is an area of large uncertainty, requiring more measurements across Scotland.

POC losses from eroding peat are considerable and in Scotland can equate to a loss of around 2 cm of vertical loss per year from bare peat gullies (Birnie 1993). As peat erodes, it enters waterways; POC losses and have been quantified from headwater streams in Northern England as between 2.2-2.8 t $CO_2$ eq $ha^{-1}$ $yr^{-1}$ (Pawson *et. al.,* 2012). There are less published equivalent measurements of POC losses from peat-dominated headwater streams in Scotland (Dawson *et. al.,* 2002) but it is clear that rivers with peat soil within their catchments such as the River Dee in Aberdeenshire contain a significant amount of POC (Dawson *et. al.,* 2012). As POC moves through the stream and river system, it is chemically and biologically processed, resulting in a further loss of carbon to the atmosphere. Current best estimates are that 70 % of POC that leaves the peatland environment is converted to $CO_2$ (Evans *et. al.,* 2013, Evans *et. al.,* 2016). This said, the conversion factor from POC to $CO_2$ remains one of the largest uncertainties in carbon budgets and a priority for future work. POC fluxes are particularly high in eroding blanket bogs where they represent one of the most important carbon loss pathways (Evans *et. al.,* 2013).

### 6.3.4 Afforestation of peat soils

Around 17 % of deep peats (that is soils with organic surface layers greater than 100 cm thick) have been forested in Scotland (Vanguelova *et. al.,* 2018). The net GHG balance of forestry on peat is difficult to calculate because of the lack of whole ecosystem GHG flux data, however measurements surveys of soil carbon stocks show a loss 30 years after plantation (Vanguelova *et. al.,* 2018). Furthermore, ecosystem carbon models suggest that planting on deep peat will causes a large loss of carbon (Matthews *et. al.,* 2020). Water loss associated with the drainage activities in preparation for forestry and uptake of water by trees can be a major source of GHGs to the atmosphere. Beyond the impact of drainage on water levels and GHG fluxes (Evans *et. al.,* 2021), gaseous losses of $CO_2$ and $CH_4$ can be significant from forestry ditches (Peacock *et. al.,* 2021). Beyond the impacts of physical disturbance and drainage of peatland, evidence associated with soil biological activity associated with trees is growing; Hermans *et. al.* (2022) found that decomposition rates were elevated in the presence of roots and Defrenne *et. al.* (2023) found that specific groups of tree-associated fungi have a major role to play in peat decomposition.

### 6.3.5 Grassland and cultivation on peat soils

It is estimated that Scotland has around 8,181 ha of cropland on peat soils and over 15,000 ha or intensively or extensively managed grassland on peat soils (Evans *et. al.,* 2017). Despite their relatively small area, these land uses have the highest emissions factors of any peat soil and are of high priority for more data collection. There are currently no measurements of carbon losses from Scottish grasslands/agriculture on peat- data need to be applied from Southern English, Northern German or Dutch study sites where there has been significant drainage for agriculture on peat soils. (Evans *et. al.,* 2017), therefore there is high uncertainty as to their carbon loss rates.

### 6.3.6 Degradation of Peaty soils

Peaty soils are primarily constituted of peaty gleys, peaty gleyed podzols and peaty podzols where the surface organic layer is less than 50cm thick) and, together they cover 26 % of Scotland and store over 800 Mt of carbon (Rees *et. al.,* 2018). They are under a number of pressures, and potential soil degradation (from the point of view of carbon), including muirburn, tree planting, and over grazing.

Tree planting on peaty soils 50 cm can reduce soil carbon stocks in the first 30 years of forestation but it is unclear what the causal factors are; drainage and soil preparation could be a major source of loss (Vanguelova *et. al.,* 2018) but as could biological, tree-driven processes (Friggens *et. al.,* 2020). Friggens *et. al.,* (2020) found that planting of birch with minimal soil disturbance caused a loss of soil carbon equal to the gain in carbon stored by the growing trees. Furthermore, natural regeneration of a mixture of native trees was found to also have significantly lower soil carbon stocks than open moorland (Warner *et. al.,* 2021). There is continuing debate over whether tree planting on peaty soils is a net carbon sink or source and what the role of disturbance through ground preparation is (Friggens *et. al.,* 2020, Smyth 2023).

Although muirburn clearly loses above-ground vegetation carbon and a long burn rotation is encouraged, it is unclear whether there is an impact on soil carbon stocks (Chapman *et. al.,* 2017). In a 50 year field experiment, it was found that burning had no significant impact on total soil carbon stocks (Ward *et. al.,* 2007). Equally, grazing did not impact total soil carbon stocks either (Ward *et. al.,* 2007). Modelling work and empirical data from a long-term herbivore density experiment on a peaty soil with a Molinia grassland suggests that commercial stocking rates of sheep will decrease soil carbon stocks and suggests lower density sheep grazing (Smith *et. al.,* 2014). Across Scotland it was shown that herbivore removal increases upland soil carbon storage (Smith *et. al.,* 2015). The impact of herbivory on soil carbon is therefore highly dependent on the soil and vegetation type and other interacting factors such as pollution (Smith *et. al.,* 2015).

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# 7 Appendix 2 – Methods

## 7.1 Assessment of soil compaction

### 7.1.1 Indicators of soil compaction

Compaction arises from the mechanical compression or deformation of the pore network in soil, so most indicators of soil compaction either measure the packing of the soil (density or porosity) or mechanical properties that measure past stress history (Chamen *et. al.,* 2015). Simply, compaction is an increase in the bulk density of the soil (mass of soil per unit volume) and generally a loss in continuity of larger soil pores (macropores) which are the main conductors of water through the soil. Both changes in bulk density and macroporosity have been used in the past to assess whether a soil is compacted or not.

The draft EU soil monitoring scheme has set critical limits for subsoil bulk densities that lead to restricted root growth and increased run off (European Commission, 2023). The critical thresholds for determining subsoil compaction are based on USDA soil quality indicators (USDA, 1999) developed further by Nyeki *et. al.* (2017) to estimate thresholds for cereal production and with the addition of bulk densities for ideal crop growth by soil texture class. However, no critical bulk density thresholds for topsoil compaction were given, possibly because topsoil compaction can be remediated to some extent through cultivation, whereas subsoil compaction is more difficult to rectify.

As bulk density is influenced by the soil texture and by the organic matter content (which has a lesser particle density), determining critical limits for topsoil bulk densities becomes more difficult. As such, Keller and Hakansson (2010) developed an equation to predict a reference bulk density that reflected how a topsoil would compact based on its sand, silt, clay and organic matter concentrations. Where a measured bulk density exceeds this value, the soil was deemed to be compacted.

Measures of soil porosity have also been proposed as a means to determine critical limits of soil compaction based on the potential for hypoxia (lack of oxygen) to occur within the soil and thus restrict root growth and development. Pores greater than 60 µm (macropores) tend not to hold water against the pull of gravity (known as the Drainable porosity or Air Capacity) and are critical for sufficient aeration to allow root growth. If the proportion of macropores fall below a critical threshold, then the soil can become anaerobic. The Air Capacity of a soil can be calculated by subtracting the volume of water held in pores at Field Capacity (notionally at -50 cm pressure head) from the volume of water held in pores at saturated water content at Field capacity.

Carter (1988) suggested that this critical Air Capacity value ranged from 8–14% Marshall and Holmes (1979, p270) da Silva *et. al.* (1994) suggested 10% and Horn (2019) suggested that this critical limit to be around 8-10%. Grable and Siemer (1968) quoted 12-15% based on their research on maize in the USA. More recently, Valentine *et. al.* (2012) showed that barley root elongation was restricted by small Air Capacity values in a range of Scottish soils under arable cultivation. Valentine *et. al.* (2012) suggested that Air Capacities <13.9% were likely to be a factor in limiting the rate of barley root elongation in soil cores taken from arable fields across Scotland.

Therefore, the critical value for Air Capacity can vary with crop type and some researchers assumed a different definition of macropores. In order to estimate the extent of compaction based on measured Air Capacity data a threshold value of 10% was selected as indicative of soil compaction with Air Capacities below this value having an impact on root development and yield through limited aeration, and in reducing infiltration rates and increasing the potential for surface runoff.

Clearly there is a relationship between soil bulk density and the Air Capacity as increasing bulk densities tend to reduce the volume of the larger pores in the soil. Data from Scottish soils shows a weak positive correlation between bulk density and Air Capacity while da Silva *et. al.* (1994) showed a clear linear relationship between bulk density and air-filled (drainable) porosity in Canadian soils.

### 7.1.2 Assessment of the extent of compaction based on Scottish data

As previously mentioned, the size and weight of agricultural machinery have increased since the 1960s (Keller *et. al.,* 2019) therefore it was decided to limit the datasets used in the analyses of soil compaction to some of the more recently collected data including the National Soil Inventory of Scotland (Lilly *et. al.,* 2011), East of Scotland

Arable Farm Survey (Valentine *et. al.,* 2012), Soilbio (Loades pers comm) and Glensaugh Farm Grid Survey (Lilly, pers comm).

Each of these datasets contained data on Air Capacity (defined as the volume of pores > 60 µm), sand, silt, clay and organic matter content and most had soil drainage class. Where this latter parameter was not available, spatial overlay with the Soil Map of Scotland (Partial cover) (Soil Survey of Scotland Staff, 1970–87) was used to identify the soil drainage class. This allowed each of the samples to be classified according to their Topsoil Compaction Risk (Lilly and Baggaley, 2018) and so to determine the proportion of Scottish soils that are potentially compacted in each risk class. Only the National Soil Inventory dataset had information on subsoils.

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## 7.2 Crop modelling

### 7.2.1 Soil Compaction Impact on Spring Barley Yield

Soil compaction involves changes in physical properties of the soil such as bulk density and soil porosity which alter the soil hydraulic and chemical properties of the soil and associated soil water and nutrient flow. The soil compaction in cultivated lands affects mostly the upper layer of the soil (topsoil compaction) but it is also observed at certain depth (subsoil compaction).

The effects of soil compaction on plant growth are complex but reductions in yield have been reported. This has frequently been attributed to mechanical impedance of root growth resulting in reduced water and nutrient uptake or insufficient aeration. Water availability for crops might be further reduced by decreased infiltration (increased runoff), resulting from the decline in hydraulic conductivity which generally accompanies compaction.

The aim of the current study was to investigate the soil compaction effects on the growth of spring barley and other crop growth factors (soil water, nitrogen, etc.) using crop model simulation across Scotland.

### 7.2.2 Material and methods

The modelling of Spring barley in this project has been undertaken using the Decision Support System for Agrotechnology Transfer (DSSAT) software platform (Hoogenboom *et. al.,* 2018), using the CERES barley model, applied with a high spatial granularity to simulate yield and other crop growth factors (soil water, nitrogen, etc.).

Unique combinations of weather and soil conditions are generated across the arable areas of Scotland (Figure 17) for use in the model by combining high resolution weather data (1 km) from UK Met Office (Met Office *et. al.,* 2018), the 1:250,000 scale National Soil Map of Scotland (Soil Survey of Scotland Staff, 1981) and data from the Scottish Soils Knowledge and Information Base (SSKIB). Only locations where barley has previously been grown during the last 12 years were considered. The locations on which barley has been grown are derived from field level reporting of land use by farmers in their annual Integrated Administration and Control System (IACS).

**Figure 17 Known barley cropping area between 2003 and 2015 the model has been applied. The white areas indicate where barley has not been grown during the last 12 years.**

### 7.2.3 Climate Data

Observed climate data were obtained from the UK Meteorological Office. The data consisted of interpolated daily data set for precipitation, maximum and minimum temperature at 1 km grid cell resolution. Daily solar radiation values for the period 1960 to 1993 were estimated using an M5 model which was developed for Scotland as a function of precipitation and temperature and the daily solar radiation values for the period 1994 to 2019 were purchased from SolarGIS (www.solargis.com) and re-sampled at 1 km grid cells to match the resolution of the other climate variables.

In total 27,997 1km grid cells were used to cover the whole barley cropped area over Scotland. The spatial pattern of the cumulative precipitation for the spring barley cropping season (between sowing and harvest) are given in Figure 18. On average, the seasonal precipitation ranged from 198 to 670 mm and can reach up to 1100 mm in some areas.

### 7.2.4 Soil hydraulic properties

To represent soil compaction, a compacted soil bulk density (BDref) was estimated for the topsoil of each of the dominant Soil Series in the soil map units using pre-established equations from the literature (Keller and Hakansson, 2010). The equation used is a function of the soil particle size and organic matter concentration and the calculated BDref represent a value for a compacted topsoil. The bulk density of topsoils deemed to have the 'optimum' bulk density for crop growth was estimated as being 87% of BDref (Keller and Hakansson, 2010). A similar approach was used to estimate the bulk densities of a compacted and non-compacted subsoil. Using data published by Nyeki *et. al.* (2017) reference values for compacted subsoils where root growth would be restricted were derived based on soil texture class along with 'Ideal' bulk densities for crop growth (Nyeki *et. al.,* 2017).

These bulk densities were then used to estimate the soil hydraulic properties (soil water content at saturation, field capacity, wilting point and the saturated hydraulic conductivity) for both compacted and uncompacted soils using pedotransfer functions (Hollis *et. al.,* 2008; Wösten *et. al.,* 1999). The soil inputs used by the model were then prepared for each unique Soil Series (228 soil series in total).

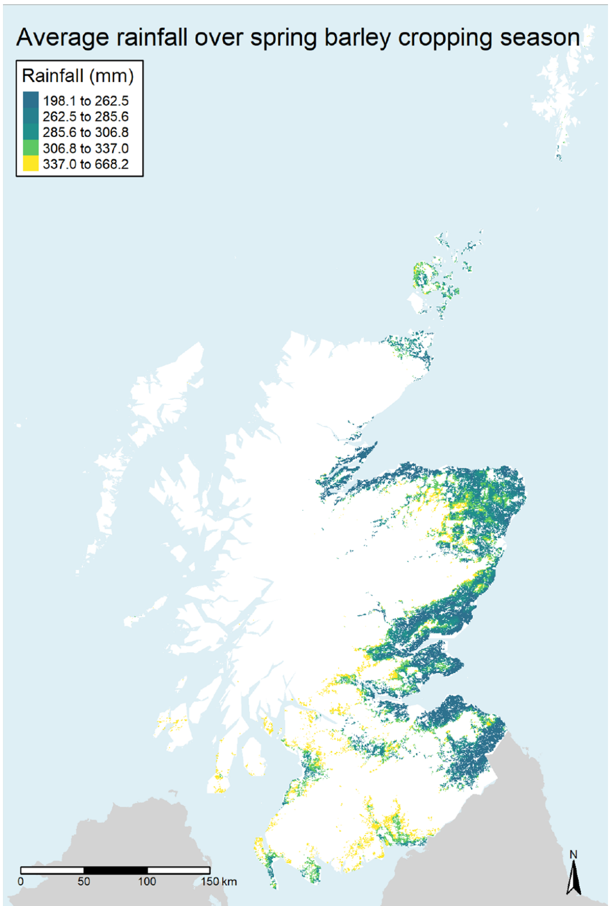


**Figure 18 Spatial distribution maps of the average growing season cumulative precipitation.**

Soil compaction affected the soil hydraulic properties and associated soil water flow. Soil water retention and transport properties are altered in response to changes in pore space geometry. This resulted in a reduction of the soil water holding capacity (SWHC) which means less water is available for crop growth. The reduction of SWHC was spatially variable and ranged between -30 and -5 mm with a median value of -13 mm. The 25th and 75th percentiles were -16 and -11 mm, respectively (Figure 19).

### 7.2.5 Barley Yield Simulation

The modelling of spring barley has been undertaken using the Decision Support System for Agrotechnology Transfer (DSSAT) software platform, using the CERES barley model to simulate yield and nitrogen. DSSAT has been satisfactorily calibrated for barley in Scotland and the same calibrated parameters were used in this study. The model inputs were prepared for each single unique combinations of the high-resolution weather grid and soil series. There are 55,088 combinations generated for locations where barley has been grown between 2003 and 2015.

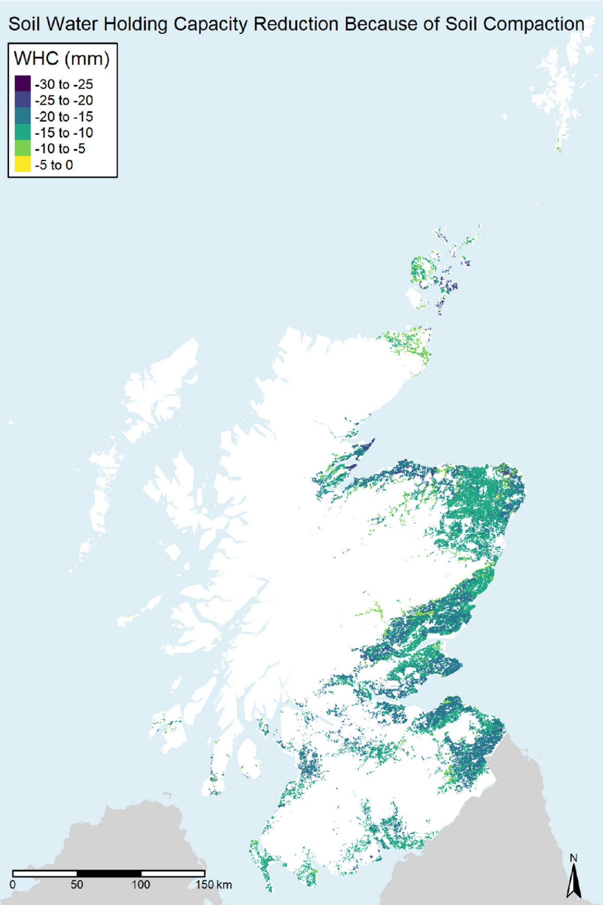


**Figure 19 Reduction of the soil water holding capacity due to soil compaction.**

No site-specific observations are available for the currently practiced sowing dates by farmers, the simulation was conducted using an early sowing date (1st of March) across the whole of Scotland. Barley yield and growth factors were then simulated for each year of the historical period 1960 to 2019. It is worth noting that sowing date can have a significant impact on spring barley yield and hence on the magnitude of the effect soil compaction may have on yield.

Since soil compaction has a direct impact on the soil hydraulic and chemical properties, two scenarios of yield estimates were considered. In the first scenario, the water limited yield potential which is the maximum yield a crop can achieve when only water is a growth limiting factor was simulated. Any yield decrease would be then attributed to increased drought due to a decrease of the soil water holding capacity and an increase in water loss by drainage and runoff so less water available for crop growth. In the second scenario, water and nitrogen limited yield was simulated when both water and nitrogen could be limiting crop growth depending on the weather and soil conditions. A one-time application of 120 kg N/ha was applied at sowing. Any additional decrease in yield as compared to scenario one would be attributed to decreased accessibility of nutrients, and increased loss of the soil nutrients by leaching, runoff, and gaseous losses to atmosphere because of soil compaction and/or the interaction of both water and nitrogen stress.

### 7.2.6 Results

Yield values are modelled for each unique combination of climate and soil where spring barley had been grown in a 1 km climate cell (Figure 17). A range of maps are presented. For the crop yields and crop growth factors, the values do consider the interactions of weather, soils, crop genetic coefficients and management in determining growth, including limitations such as drought or leaching of nutrients that limit growth. The crop model simulations do not reflect yield losses such as those from wind or pest and disease damage. Penetration resistance of roots into soil due to mechanical impedance is also not accounted for.

### 7.2.7 Soil Compaction Effect on Water Limited Yield Potential

There is a large range in spatial and temporal variation in the barley water limited yield potential (Yw) estimates for the period 1960-2019 (Figure 20) for the uncompacted soils. Average yield ranged between 1.5 to 9.0 t/ha, which corresponds to the maximum yield that could be achieved if nitrogen was not limiting. The national average yield potential is estimated at 7.6 t/ha with an average coefficient of variation of 18.0% across the barley cropped areas. Areas of high estimated yield potential occur due to favourable growing conditions, primarily through combinations of good soil water retention (and/or slow drainage) resulting in lower water stress to crops and favourable weather conditions at key growth stages, particularly adequate rainfall, especially in the spring. Similarly, areas of low yield occur due to unfavourable conditions of poor water retention (higher risk of water stress), such as soils with a high sand content; rapid drainage or high run-off and unfavourable weather conditions (primarily low rainfall and warmer temperatures leading to higher evapotranspiration rates and soil water loss). Areas with a high coefficient of variation occur due to high yearly variability in weather conditions mainly fluctuations of precipitation and soil with low soil water holding capacity. Areas with relatively low average yield

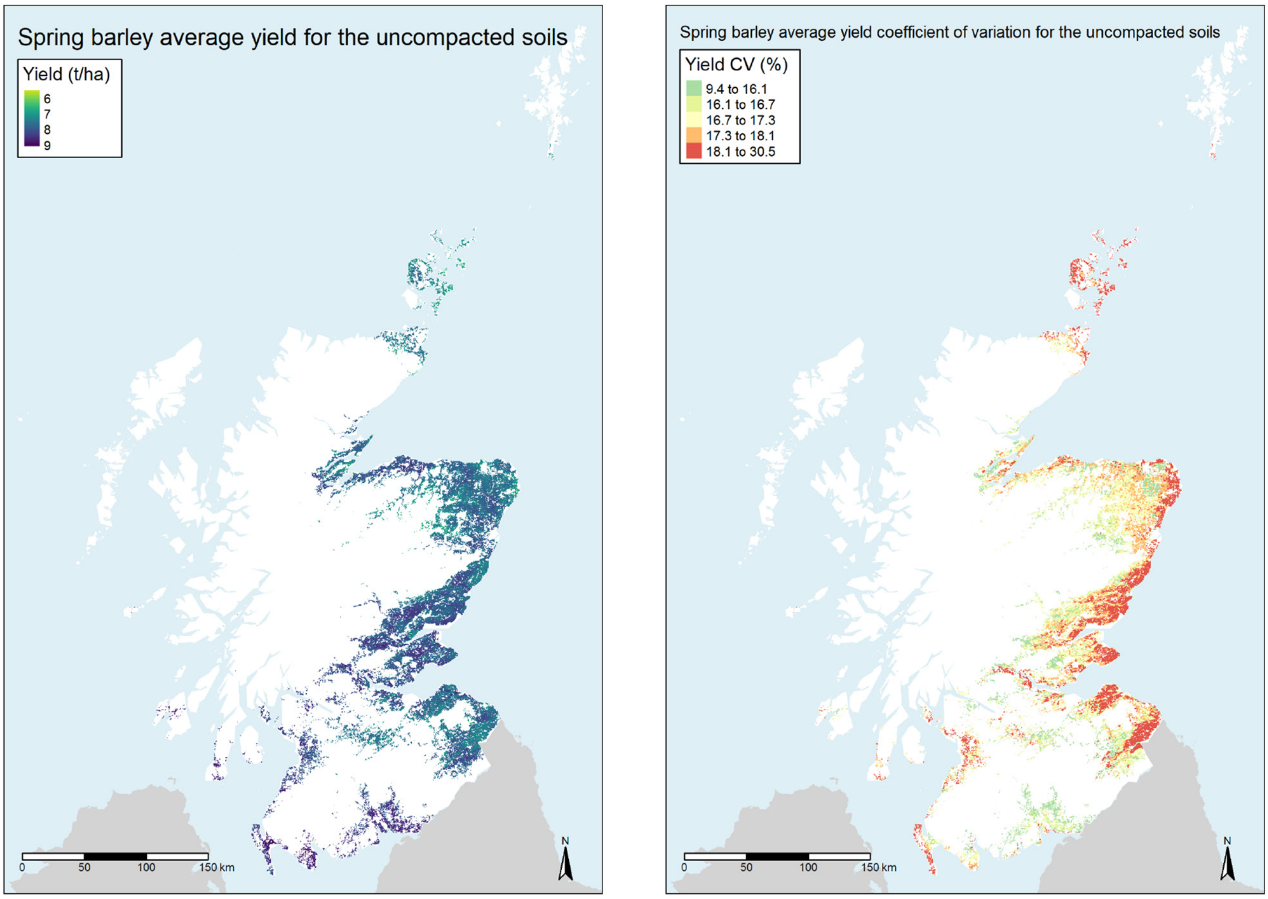


**Figure 20 Spatial variation of spring barley water limited yield potential average (left) and coefficient of variation (right) over the period 1960-2019 for the uncompacted soils.**

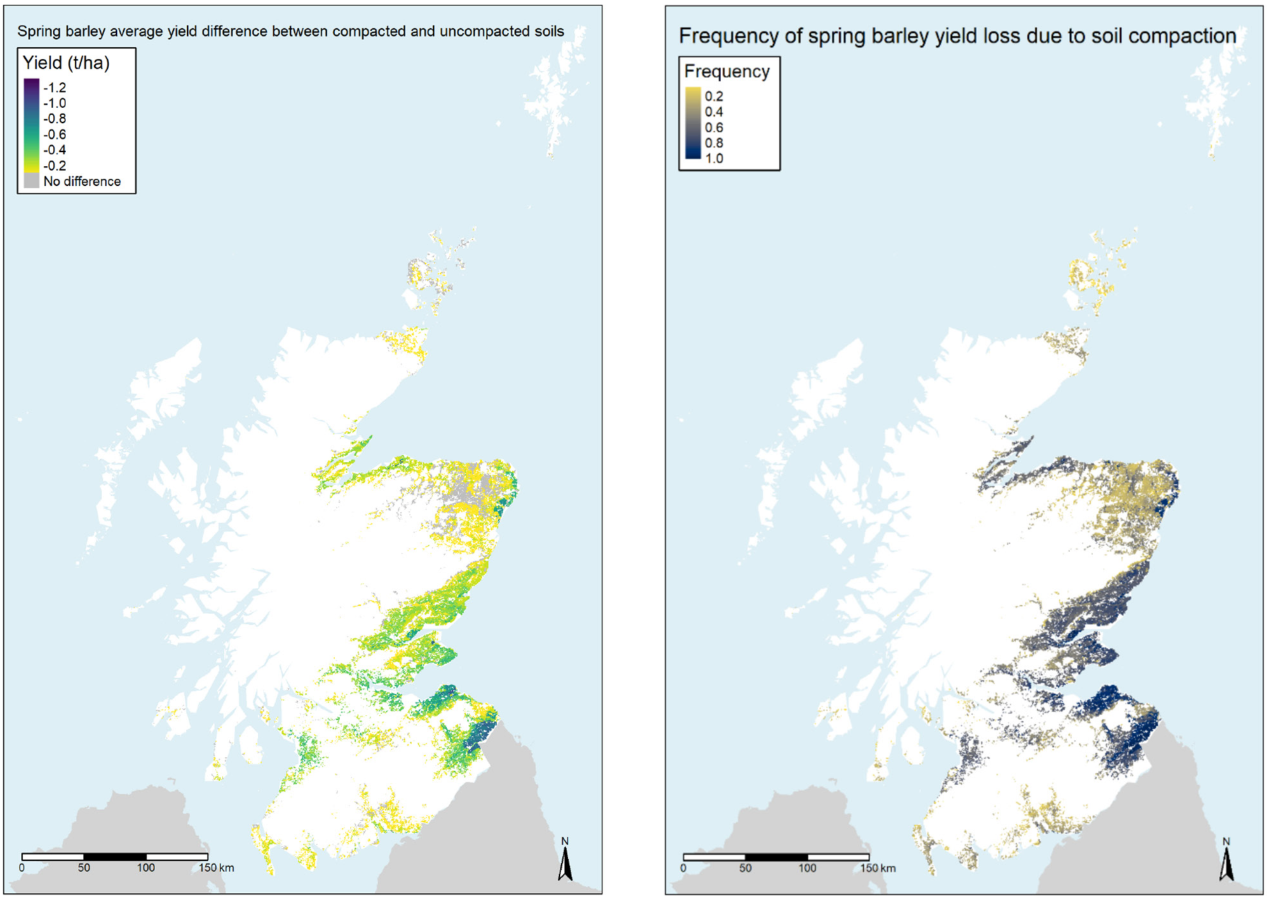


**Figure 21 Spatial variation of spring barley average water limited yield potential loss (left) and frequency of yield loss (right) over the period 1960-2019 due to soil compaction.**

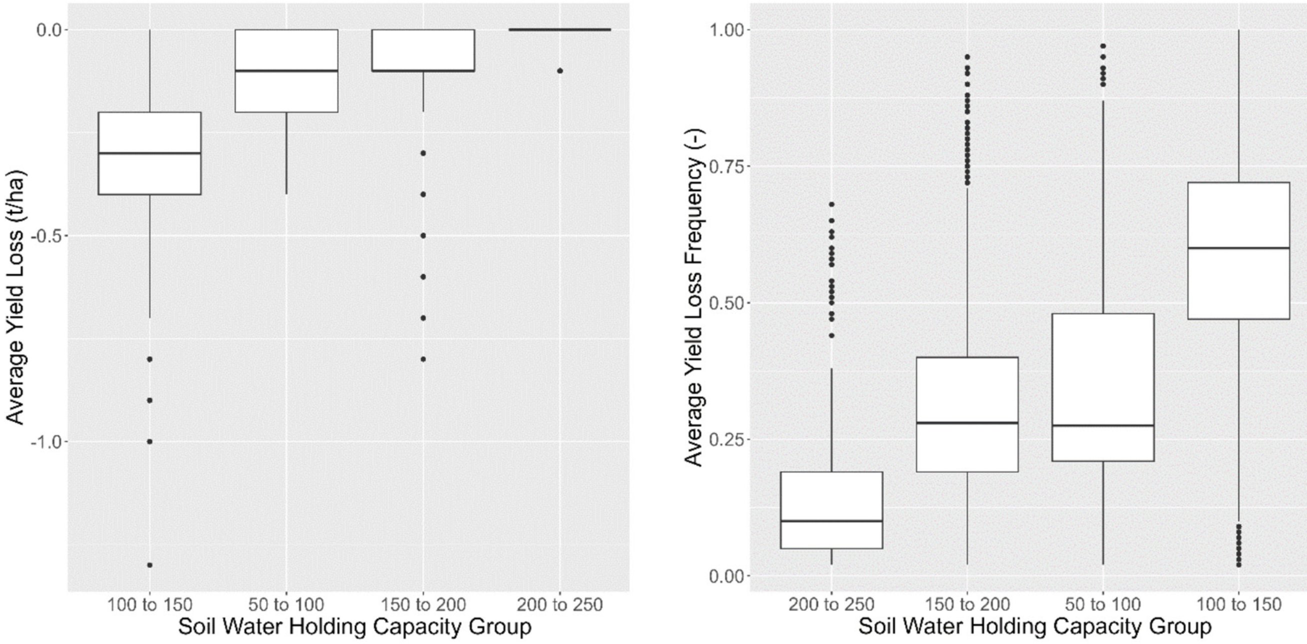


**Figure 22 Average water limited yield loss (left) and average yield loss frequency (right) due to soil compaction by soil water holding capacity group.**

have generally a high coefficient of variation.

The overall effect of soil compaction on barley yield was negative for the water limited yield potential (Figure 21) because of the effect soil compaction had on the soil water holding capacity but it also resulted in few cases in no effect depending on the soil hydraulic properties and the amount of precipitation during the cropping season. The absolute yield loss was calculated as the difference between years for the compacted and uncompacted soils. The average absolute yield loss during the period 1960 to 2019 is spatially variable and ranged between 0 to 1.3 t/ha (0 to 18.2 % of barley yield). The highest yield loss was observed for the soils with a low soil water holding capacity (between 100 and 150 mm) and a cumulative precipitation over the cropping season of less than 250 mm. The highest frequency of yield loss was also observed for these conditions (Figure 22). Conversely, spring barley grown on soils with a high soil water holding capacity was not affected by soil compaction.

## 7.2.8 Soil Compaction Effect on Water and Nitrogen Limited Yield

When nitrogen is considered as an additional growth limiting factor, the additional average yield losses due to nitrogen losses and/or the interaction between both water and nitrogen stresses ranged between -6 and 0% (Figure 23). There is a large range in spatial and temporal variation in the additional barley yield losses for the period 1960-2019 with a median loss value

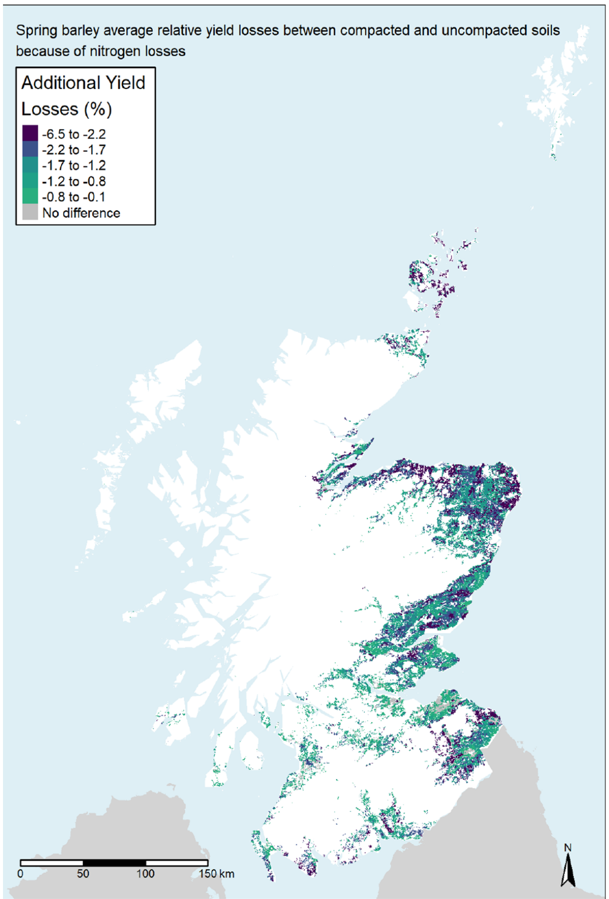


**Figure 23 Average water limited yield loss (left) and average yield loss frequency (right) due to soil compaction by soil water holding capacity group.**

of 1.3%. The 25th and 75th percentile losses were 2.0 and 0.8%, respectively.

The additional yield losses were mainly due to slightly increased nitrogen leaching and a slight increase in nitrous oxide emissions (Figure 24)

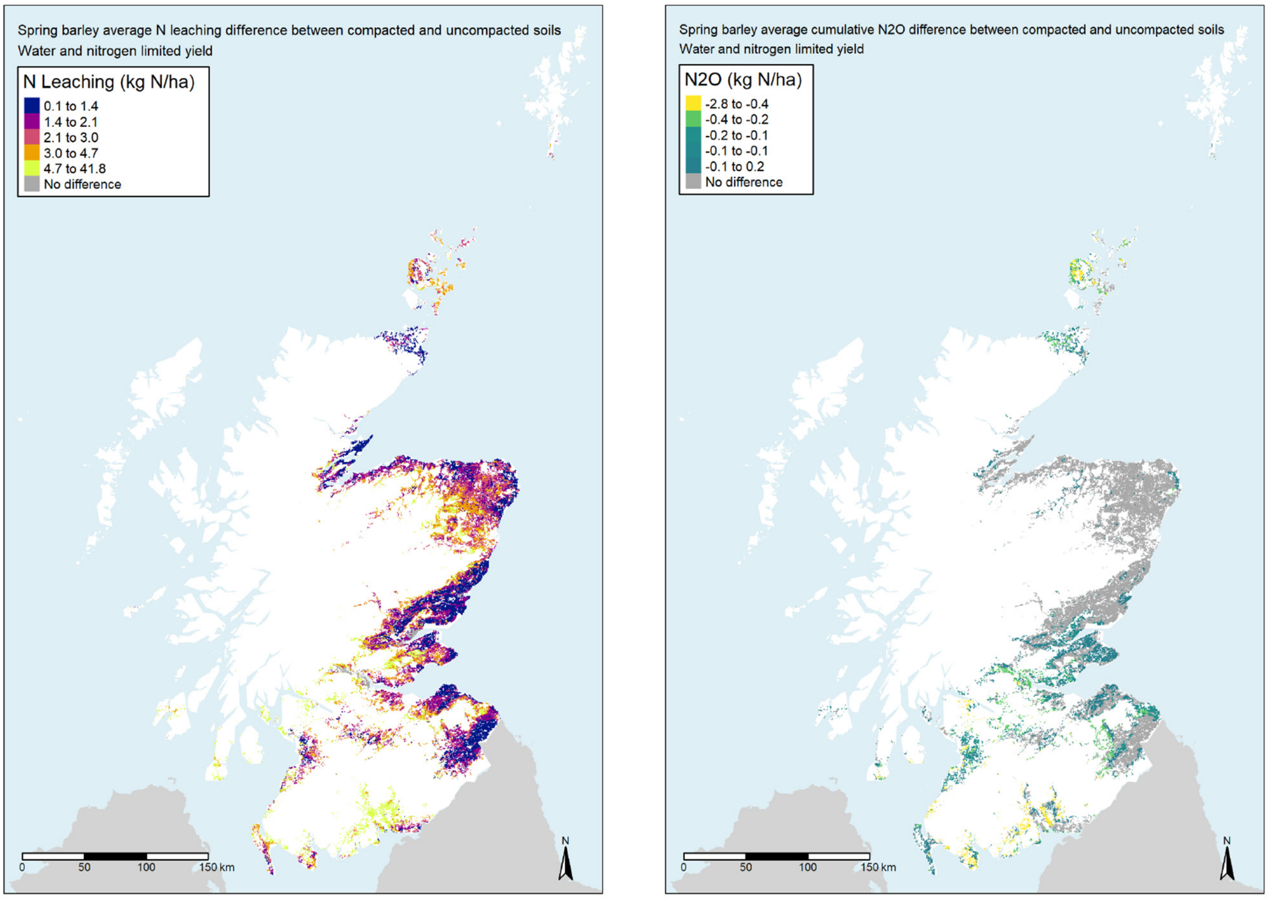


**Figure 24 Spatial variation of the average nitrogen leaching (left) and nitrous oxide emissions (right) difference between compacted and uncompacted soils.**

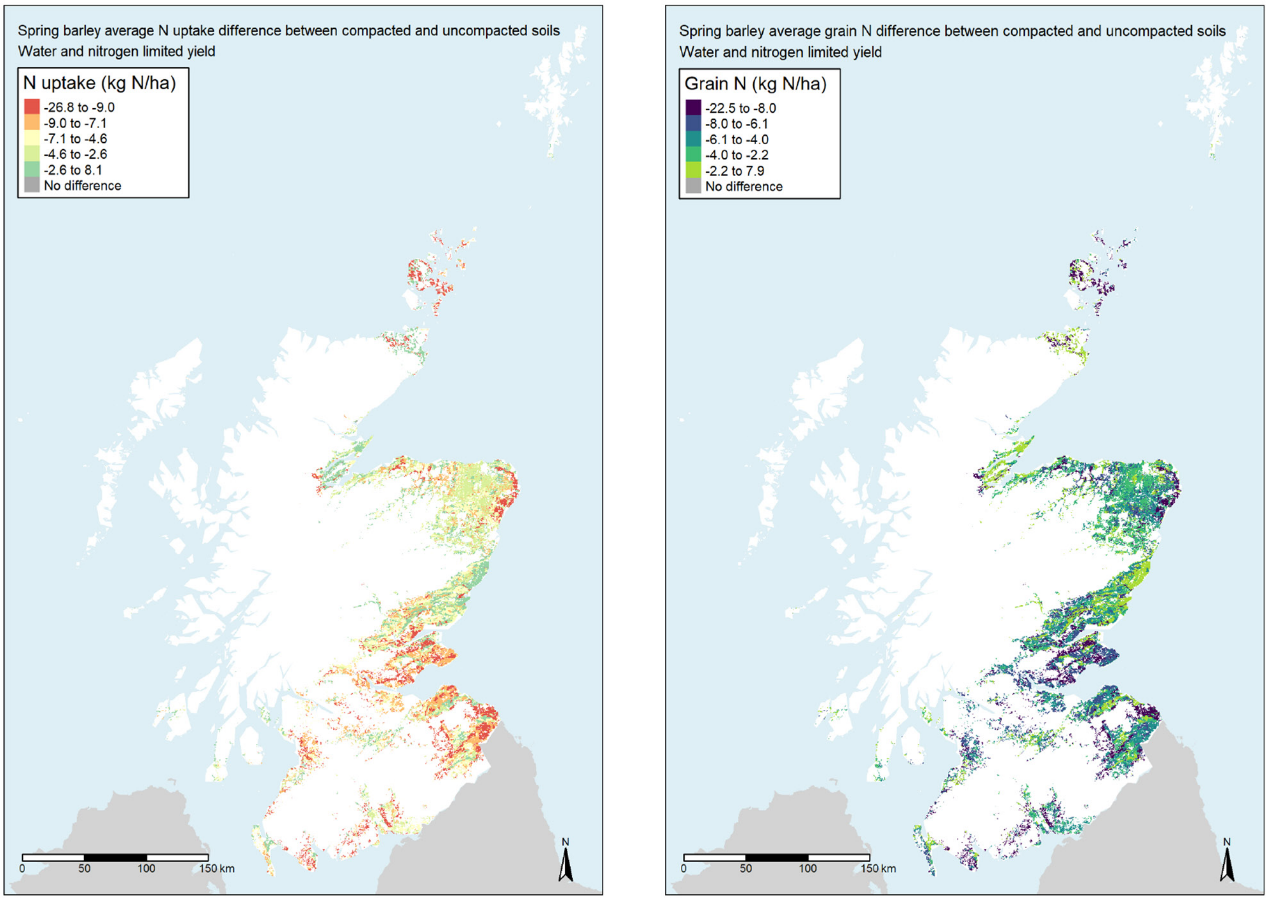


**Figure 25 Spatial variation of spring barley average nitrogen uptake (left) and grain nitrogen content (right) difference between compacted and uncompacted soils.**

resulting in a decrease in nitrogen availability to crop growth. This led to a decrease in nitrogen uptake which had a direct impact on grain nitrogen content and thus grain quality (lower grain nitrogen content).

Soil compaction reduced crop nitrogen uptake by about 5.8 kg N/ha on average. The reduction in nitrogen uptake was spatially and temporally variable and ranged between -26 and 0 kg N/ha. The barley locations with the highest reduction in N uptake had the highest reduction in yield due to soil compaction. The reduction in N uptake had a direct impact on grain nitrogen content which was reduced by 5 kg N/ha on average. The reduction in grain N content varied between -23 and 0 kg N/ha (Figure 25).

### 7.2.9 References

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# 8 Appendix 3 – Infographic

## ASSESSING THE SOCIO-ECONOMIC IMPACTS OF SOIL DEGRADATION ON SCOTLAND'S WATER ENVIRONMENT

Healthy soils
are important to the Scottish economy
improving crop yields, storing water to help limit
the impacts of drought and storms, and regulating
water flows to rivers and lochs.
Degraded soils have both direct and indirect costs
to individuals, society and the wider economy.

### COMPACTED SOILS

YIELD LOSS ESTIMATED AT

**£16-49 MILLION PER YEAR**

### COMPACTED SOILS

ADDITIONAL FUEL USE FOR FIELD OPERATIONS

**£9-26 MILLION PER YEAR**

### COMPACTION AND SEALING

INCREASED FLOOD RISK AND INSURANCE CLAIMS

**£57k-76k PER HOUSEHOLD CLAIM**

### SOIL CONTAMINATION

LOSS OF LAND, DEGRADED WATER & FOOD QUALITY

**£ NOT YET ABLE TO CALCULATE**

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